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A REVIEW OF HYDROGEOLOGICAL AND GEOCHEMICAL DATA FROM THREE ONTARIO HYDRO FLY ASH DISPOSAL SITES

Report No 89-42-K

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ABSTRACT

A review of hydrogeological and geochemical data collected from three Ontario Hydro fly ash disposal sites has been completed. This review was conducted, as part of the Lakeview Ash Management Study, for the purpose of assessing the environmental impact of fly ash landfilling on the groundwater regimes at the following sites: 1) Booth Quarry in Etobicoke, 2) Canada Brick Quarry in Mississauga, and 3) Birchwood Park in Clarkson, Ontario. Fly ash has been in place at the three above sites from 3 to 17 years, and different landfill designs and practices were used at each one. Geologically, the Booth Quarry and the Canada Brick Quarry are both located in shale bedrock which is a favourable environment for fly ash landfilling. The Birchwood Park site, located in a former sand and gravel pit, has not been as successful in terms of containment of major fly ash leachate constituents. Based on the data collected to date, no significant degradation of local groundwater quality has taken place, however, continued monitoring of hydraulic heads and water quality is recommended until equilibrium conditions have been reached. The use of fly ash as a structural fill material has allowed for the reclamation of land for recreational and developmental programs in the large urban centres of the Toronto area.

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EXECUTIVE SUMMARY

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A review of hydrogeological and geochemical data collected from three Ontario Hydro fly ash disposal sites has been completed. This review was conducted, as part of the Lakeview Ash Management Study, for the purpose of assessing the environmental impact of fly ash landfilling on the groundwater regimes at the following sites: 1) Booth Quarry in Etobicoke, 2) Canada Brick Quarry in Mississauga, and 3) Birchwood Park in Clarkson, Ontario.

Fly ash from Lakeview TGS has been in place at the above three sites for 10, 3 and 17 years respectively and different landfill designs and practices were used at each one. For each of these sites, the following aspects were reviewed: i) specific design details and history of landfilling, ii) overburden and bedrock stratigraphy, and iii) all available hydraulic head and geochemical data.

The Canada Brick Quarry, located in shale bedrock, is the youngest ash landfill examined during this study. The site has been extensively instrumented with piezometers, to determine background and on-site groundwater conditions and to monitor any changing conditions down-gradient of the site. Monitoring has been underway since landfilling began in 1981. No degradation of local groundwater quality, through fly ash leachate contamination, has been detected to date. However, the ash fill has only recently been capped and "in-ash" water levels have not approached equilibrium conditions. Geochemical monitoring at this site must be restricted to analyzing for minor and/or trace elements (boron and arsenic) since the major ion chemistry (calcium, sodium, sulphate) of the groundwater in the shale bedrock is similar to that of ash leachate.

The Booth Quarry, also located in shale bedrock, was landfilled with fly ash between 1976 and 1979. Four shallow piezometers and the sump within the ash fill have been monitored for hydraulic heads and sampled for water chemistry. To date, no change in background water chemistry has been detected immediately down-gradient of the quarry. However, only recently (Summer/88) has the water

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level within the ash fill risen up to equilibrium conditions with the surrounding groundwater table. Up to this point, local groundwater flow would have been towards the quarry, thus preventing any significant transport of fly ash leachate into the surrounding shale bedrock. Two aspects of the present monitoring program need improvement: 1) fly ash leachate chemistry should be determined from a new sampling point since it appears that the sump water no longer represents in situ ash leachate, and 2) multilevel monitoring devices installed immediately down-gradient of the quarry would allow for sampling and analysis of groundwater along the entire depth of the ash landfill (12 metres).

The Birchwood Park site, formerly operated as a sand pit, was landfilled with Lakeview TGS fly ash between 1968 and 1972. The landfill design did not incorporate the use of a clay liner even though the host medium consisted of permeable silts and sandy silts. No leachate collection system was provided which would have permitted for control of leachate generation and transport. Two hydrogeologic studies were completed at this site: the first one between August 1978 and June 1980, and the second one between January 1985 and September 1985. This study revealed that a contaminant plume of elevated major ions such as calcium, sodium and sulphate had developed below and down-gradient from the ash fill. With time, this plume has thinned out and migrated approximately 50 m down-gradient. The migration of minor leachate components, such as boron and strontium, and trace leachate components, such as chromium and selenium, has been mainly restricted to a zone below the ash fill. Proper landfill design methods, restricting the generation of fly ash leachate at Birchwood Park, would have substantially reduced the size of the contaminant plume.

In conclusion, no off-site migration of fly ash leachate has been detected at any of the three landfill sites examined during this study. At present, local groundwater at all three sites is not being used as water supplies. In fact, the natural groundwater in the shale bedrock at the Canada Brick and Booth quarries is unfit for consumption since the concentrations of several chemical constituents (ie chloride and sulphate) are above the drinking water standard. These natural concentrations are sometimes higher than those found in fly ash leachate.

Geochemical monitoring should continue at all three sites to provide additional data for future environmental impact assessments. It is recommended that new monitoring points should also be installed within the ash at both Canada Brick and Booth quarries to obtain adequate analyses of in situ fly ash leachate chemistry.

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A REVIEW OF HYDROGEOLOGICAL AND GEOCHEMICAL DATA FROM THREE ONTARIO HYDRO FLY ASH DISPOSAL SITES

1.0 INTRODUCTION

As part of the Lakeview GS Ash Management Study, a review of data collected from three of Ontario Hydro's fly ash landfill sites has been conducted. The landfill sites selected for review are all near the Lakeview Generating Station where the majority of the fly ash used in these projects was produced. These sites include two former shale quarries, Booth Quarry in Etobicoke and Canada Brick Quarry in Mississauga, and one former sand and gravel pit, Birchwood Park in Clarkson.

The principal objective of this review is to assess the environmental impact of fly ash landfilling on the groundwater regime at the above sites. The procedure to be followed is: i) to briefly review the history of landfilling of each site, including any specific design details of interest, ii) to detail the overburden and bedrock stratigraphy, iii) to review all available hydraulic head data to determine groundwater flow directions and gradients, and note any changes which may have occurred as a result of landfilling, and iv) to review all available geochemical data, including major, minor and trace element concentrations and to determine if the fly ash leachate has migrated away from the landfill site.

Each site selected possesses some interesting characteristics which make them important from an ash management point of view. The Booth Quarry is the oldest fly ash disposal site located within the shale bedrock of the Toronto area. Landfilling operations were completed by 1979 and therefore a study of this site should allow an assessment of the choice of shale bedrock as a containment medium for fly ash. The Canada Brick fly ash disposal site is also situated in the same shale bedrock of the Georgian Bay Formation, but fly ash landfilling was only completed at the end of 1986. The fly ash has been capped with a clay cover and quarrying operations continue at other locations within the company property. Although this site is relatively new, it is also the most extensively instrumented and monitored which will help assess the impact of the initial filling operations. Previous hydrogeologic impact studies have been conducted at the Birchwood Park site where fly ash landfilling was completed in 1972. Results from these studies will also be

reviewed because this site serves as an example of early landfilling procedures and provides details on the migration of fly ash leachate in unconsolidated, higher permeability materials.

2.0 CANADA BRICK QUARRY

2.1 Background

2.1.1 Quarry Operations

Brick production at the Canada Brick site (formerly known as Domtar Quarry) started in 1911. Shale, used in the brick manufacture, has been quarried on-site since the construction of the plant. The quarry, situated on the east side of the plant, occupies an area of approximately 21 ha (Figure 1). It is bounded to the northwest by a CP railway, to the southeast by Dundas Street West and to the northeast by a residential subdivision. The quarry has been worked in 2 benches, one approximately 10 m below grade, and the other at 25 m below grade. Quarrying of the deeper shales has been limited to an area of about 8 ha because of the higher salt content at that depth.

2.1.2 Landfilling Operations

To minimize the area of ash exposed to wind, the exposed working surface was limited to 1 hectare (2.5 acres) cells (Figure 1). A total of seven cells, separated by containment berms, were landfilled with fly ash and bottom ash originating from the Lakeview Thermal Generating Station. From December 1981 to December 1986 a total of 683,927.4 tonnes of ash was placed in the quarry. Based on a target dry density value of 1.116 Mg/m^3, this corresponds to a total ash volume of 557,125 m^3. Cells 1 through 7 were landfilled up to an elevation of 116 to 117 metres above sea level which corresponds to an ash thickness of between 8 and 10 m. A seal layer with an in situ hydraulic conductivity no greater than 1×10^{-7} cm/s was constructed to cap all of the cells/1/.

An underdrainage system was constructed for each one of the cells prior to the placement of ash. The individual underdrainage systems were all interconnected to form a large network draining into a central sump located near the south edge of Cell #1. The purpose of this underdrainage system was to: 1) provide a source of recycled water for dust control, 2) control the water level in individual ash cells during compaction, and 3) control the water level in the final reclaimed quarry over the long term.

Surface drainage control was achieved by the use of deflection berms, surface grading and drainage ditches, designed to channel surface run-off to a clean water sump located up-gradient of the filling operations.

2.2 Geology

2.2.1 Overburden

The veneer of unconsolidated sediments which overlies the shale bedrock in the vicinity of the quarry, ranges in thickness from 1 to 3 m. North and east of the quarry, this overburden consists of a compact, sandy silt till. South of

the quarry and below the escarpment which transects the southeastern corner of the property, the overburden consists of sand, silt and occasional thin seams of gravel. These lacustrine deposits are associated with the shoreline of glacial Lake Iroquois which covered the area south of Dundas Street.

2.2.2 Bedrock

The bedrock encountered at the quarry is known as the Dundas shale of the Georgian Bay Formation. In the Toronto area, the formation is over 120 m thick. The shale is a moderately soft, bedded grey rock with platy laminations. About 15 per cent of this formation is comprised of limestone and sandstone interbeds which range in thickness from about 1 to 12 cm and are much harder than the shale beds. Weathering has resulted in extensive fracturing of the surficial 3 m of the formation. Below this weathered zone, fractures are limited to horizontal bedding planes and vertical joints. An oxidized zone extends to a depth of about 5 m.

2.2.3 Hydraulic Conductivity

The hydraulic conductivity of the Dundas shale in the vicinity of the Canada Brick quarry was previously determined during initial site investigations in 1981/2/. Tests performed included bail-down tests in piezometers and pressure-packer tests in open boreholes. The change in horizontal hydraulic conductivity (Kh) of the shale varies from 10^{-3} to 10^{-9} cm/s between 5 and 25 metres below ground surface. This variability is shown graphically in Figure 3.

The hydraulic conductivity of shale deposits is governed by fracture porosity and permeability. It is not surprising therefore that Figure 3 shows a large degree of scatter. The general trend of decreasing hydraulic conductivity with depth is another characteristic of shale deposits. As the overburden pressure increases, fractures tend to become closed and in addition, the inherent plasticity of the shale beds precludes the formation of fractures at great depths. From Figure 3 it may also be concluded that the majority of the groundwater flux occurs in the top 5 to 10 m of the shale deposit where hydraulic conductivities are generally greater than 1×10^{-2} cm/s.

2.3 Monitoring Program

2.3.1 Objectives

The objectives associated with installing piezometers around the Canada Brick quarry were/1/:

- to determine background, up-gradient water chemistry and groundwater flow conditions
- to determine and monitor water chemistry and flow conditions in and around the perimeter of the quarry
- to determine and monitor water chemistry and flow conditions down-gradient of the quarry, and to help assess the impact of fly ash landfilling on the local groundwater regime.

2.3.2 Piezometer Locations

A total of 37 piezometers were installed in and around the Canada Brick quarry. Since these piezometers were installed in 1980-81, a total of 15 have been destroyed to date because of local construction activities. Ground level elevations and intake bottom elevations for each piezometer are tabulated in Table I. The intake lengths for individual piezometers range from 1.0 to 2.5 m.

The piezometers may be grouped into three different categories: 1) up-gradient piezometers (2A and B, 5A and B, 6, 7), on-site piezometers (1, 3A, 3B, and 3C, 4, 8, 9, 10, A, G), and down-gradient piezometers (B, C, D, E, F, H, I, J). The locations of these piezometers are shown in Figure 1. The locations of the piezometer tips are shown in the two cross-sections of Figure 2.

2.4 Groundwater Flow Conditions

The regional groundwater flow direction in this area is towards the southeast and Lake Ontario. The hydraulic gradients associated with the vertical and horizontal components of groundwater flow can be determined by analyzing the water level data from the network of piezometers around the Canada Brick quarry (see Table II).

Based on the most recent tabulated water level data, the horizontal gradient (i_h) north of the local escarpment is 0.03 towards the southeast. In the residential subdivision, which is at a much lower elevation, $i_h = 0.02$.

Vertical hydraulic gradients (i_v) can be determined at locations where two or more piezometers are installed at different depths. Up-gradient of the fly ash fill (BH-2 and BH-5) vertical hydraulic gradients are downward and range in magnitude from 0.7 to 0.8. Periodically, the gradient has been greater than one.

Down-gradient of the fly ash fill, in the residential subdivision, the vertical hydraulic gradient changes substantially with depth. BH-C and BH-D, located on the southeast side of Dundas Street indicate that the vertical hydraulic gradient ranges between 0.6 and 0.85 downwards. The piezometer intake lengths of these two nests are all at or above the elevation of the quarry bottom. The high i_v is probably caused by infiltration through the local topographic high of the escarpment immediately to the northwest. Near the surface (111.4 m.ASL at BH-E), the vertical hydraulic gradient between BH-E(s) (109.5 m) and BH-E(i) (102.6 m), is approximately 0.33 downward. Between BH-E(i) (102.6 m) and BH-E(D_2) (88.4 m) i_v reduces to approximately 0.05 downward. At BH-F, located 225 m southeast of BH-E, the vertical hydraulic gradient near the surface has reduced to 0.01 downward and has periodically reached 0.05 upward. At BH-J, approximately 220 m northeast of BH-E, i_v has been equal to zero since installation in November of 1982. This reduction in hydraulic gradients, and corresponding groundwater velocity, is

due to the small change in slope of the ground surface towards Lake Ontario, which acts as the regional discharge area.

Some anomalous groundwater levels have been recorded in the piezometer nests located between the escarpment and the southeastern limit of the Canada Brick quarry. Water levels from BH-10 located within the gully of Mary Fix Creek, indicate that the vertical hydraulic gradient equals 0.6 downwards. However, between BH-8 (105 m.ASL) and BH-G (106.9 m), the vertical hydraulic gradient has ranged between 0.15 and 0.33 upwards since October of 1986. This is due to an increase in water level at BH-8 of between 1 and 3 m above background since early in 1986 (see Table II). The same type of increase is also noted in BH-3A (106.6 m.ASL), which occurred at approximately the same time and has persisted to date. These increases occurred in conjunction with a general increase in water level in sump #1. Whether these increases are inter-related remains unclear at this time because of a lack of any recent field response test results.

Since mid-1983, the water levels recorded at BH-3B, 3A and 3C indicate that a downward hydraulic gradient exists between 3B and 3A, and that an upward hydraulic gradient exists between 3C and 3A. Based on a set of readings taken in June of 1987, i_v = 0.7 downwards between 3B and 3A and i_v = 0.13 upwards between 3C and 3A. This upward gradient may be a result of placement of fly ash into Cell #2. This activity would have reduced the infiltration of surface water into the local shale, thereby favouring shallow, groundwater flow which would tend to discharge down-gradient of the shale quarry.

2.4.1 Fluctuations in Sump Water Level

Sump #1, located in the south corner of Cell #1, serves as the collection point for the vast underdrainage network in all fly ash cells to the north. During the landfilling operations, Sump #1 was mainly used as a source of water for spraying the landfilled fly ash. This helped in the compaction of the ash and also in the suppression of fly ash dust. Since early in 1986, the amount of water required for dust control and compaction has greatly reduced because landfilling operations were completed and a layer of cover material was applied to the fly ash. The level of water in the sump has been allowed to rise and correspondingly the water level in the ash fill has probably also risen. Since the clay cap covering the fly ash fill will restrict infiltration of water from the surface, the rise in water level within the ash will be due to groundwater inflow from the up-gradient rock mass.

2.5 Groundwater Chemistry

Groundwater quality in the vicinity of the Canada Brick Quarry landfill has been monitored on a regular basis since filling began in December of 1981. One of the conditions for the approval of the fly ash landfill by the Ontario Ministry of the Environment was that up-gradient wells be monitored annually, on-site wells four times per year and down-gradient wells three times per year. Sump #1 in Cell #1 was considered an on-site well and monitored four times per year.

The rationale for the sampling and testing program may be summarized as follows: 1) up-gradient wells monitor fluctuations in background water quality and indicate the chemistry of the groundwater that may be saturating the ash fill, 2) on-site wells allow for monitoring groundwater quality around the perimeter of the ash landfill and the sump provides "in-ash" leachate quality, 3) down-gradient wells allow for monitoring of water quality in the shale underneath the residential subdivision to the south.

2.5.1 Major Ion Chemistry

The massive amounts of data collected by the Chemical Research Department must be summarized in graphical form to allow interpretation. To this end, the Piper plot has been used to represent the major cation and anion concentrations of groundwaters sampled at several piezometers. The various ionic concentrations are plotted as percentage equivalents per million which allows for waters of different total concentrations to be represented by the same point. One of the interesting aspects of the Piper plot is that it accommodates the plotting of a large number of analyses and is also convenient for observing the effects of mixing two waters from different sources, ie fly ash leachate and bedrock groundwater. Implicitly, however, these two waters must have sufficiently different ion balances to plot in different areas of the Piper diagram.

Figure 4 is a Piper plot showing the major ion chemistry results of the sump water collected from March 1982 to June 1987. The lower left triangle represents the cation chemistry, the lower right triangle represents the anion chemistry and the central tetrahedron represents the total major ion chemistry. Although there is some scatter in this plot, the data tend to concentrate in one area of the diagram. This area of the central diagram represents water which has a dominance of alkali metal cations (Na^+ and K^+) and strong acid anions (Cl^- and $SO_4^=$).

To be able to detect the mixing of fly ash leachate with the local groundwater, the major ion chemistry of the bedrock waters must initially plot a different area of the Piper diagram. If through time the points representing the local groundwater chemistry begin plotting closer to the area represented by the ash leachate chemistry, then this may be evidence of initial contamination. One must also look for changes in specific ion concentrations and total dissolved solids.

2.5.1.1 Up-gradient Chemistry

Figure 5 is a Piper plot showing the sump water chemistry and the up-gradient groundwater chemistry represented by piezometers 2A and B and 5A and B. The up-gradient, or background, water chemistries from these piezometers are the results of sampling and testing in June 1987. Although these piezometers were sampled on a yearly basis, their major ion chemistry has not changed significantly and therefore one point has been plotted for the sake of simplicity. All of the data is available for examination in annual report #7/3/. Piezometers 2B and 5B are both located approximately 15 m below ground surface and therefore the water sampled from these points is representative of deep shale groundwater. As shown in Figure 5, the major ion chemistry of this water is the similar to the sump water since they both plot in the same area

of the central diagram. More specifically, the deep groundwater anionic chemistry is dominated by chloride as opposed to that of the ash leachate which has more of a balance between chloride and sulphate. The groundwater sampled from 2A and 5A has a significantly different anionic chemistry which is representative of waters dominated by a weak acid anion (bicarbonate). These piezometers, located at a depth of 7.5 m BGL, must be within the weathered zone of the shale bedrock where Cl^- and $SO_4^=$ have been leached away.

The immediate conclusion from examining Figure 5 is that if the major ion chemistry of the down-gradient piezometers is represented by points 2B and 5B, it will be very difficult to detect contamination by fly ash leachate based on major ion chemistry alone.

2.5.1.2 On-Site

Figure 6 is a Piper plot representing the major ion chemistry of the on-site piezometers 3A and 3B and 3C, 8 and G. These piezometers are located immediately down-gradient of the fly ash cells (Figure 1) and below the local escarpment. Once again, the major ion chemistry of all of these sampling points show similar characteristics to that of ash leachate with the exception of chloride which dominates the chemistry. Groundwater chemistries from points 3A, 3C and G are represented by a single point since their major ion chemistries have remained relatively constant throughout the sampling period. The groundwater chemistries at piezometers 3B and 8 have changed since April/84 and June/83 respectively and have been fluctuating continuously since that time. The two additional points on the Piper plot of Figure 6 (8-2 and 3B-2) are representative of this change in chemistry. Rather than being evidence of ash leachate contamination, this change appears to represent a mixing of two waters where the infiltrating water is more dominated by a weak acid anion (bicarbonate or HCO_3^-), similar to that of near-surface piezometers 2A and 5A shown in Figure 5.

The change in water chemistry at piezometer 3B and 8 may be correlated to a dramatic increase in water level (10 m) in the nearby, up-gradient piezometer #9 (Figure 1). This piezometer is located within the local escarpment which separates the Canada Brick quarry from Dundas Street to the southeast. This increase in water level occurred sometime in late 1983 or early 1984, and may have been due to local construction activities which inhibited drainage of infiltrating water from the local shale mound. The water level in piezometer #9 has remained consistently high since that time and may have caused the near-surface water to percolate towards piezometer 3B and 8 located immediately down-gradient.

2.5.1.3 Down-gradient

Figures 7 and 8 are Piper plots which represent the major ion chemistry of groundwaters below the residential subdivision southeast of the quarry. Figure 7 represents data collected from piezometers D(d), D(s), E(d), E(s), F(d) and F(s). Since the water chemistry at these piezometers has remained constant throughout the sampling period, only one point has been plotted to represent each sampling point. All the data are available for review in an annual report/3/. Once again the deep piezometers, D(d), E(d) and F(d) (10 m, 24 m and 10 m.BGL respectively) appear to have a total major ion chemistry

similar to that of ash leachate with the exception of chloride dominance over sulphate in the deep groundwaters. The shallow piezometers, D(s), E(s) and F(s) (4 m, 6 m and 5 m respectively) are within the leached zone of the shale bedrock and therefore the groundwater chemistry shows more dominance of alkali earth cations (Ca^{2+} and Mg^{2+}) and weak acid anions (HCO_3^-). The chemistry from piezometer F(s) seems to be more indicative of mixing between deep and shallow groundwaters caused by the small reversals in hydraulic gradient discussed in the previous section.

Figure 8 represents the major ion chemistry results obtained at piezometers J(d) and J(s). The groundwater chemistry at J(d) (10 m.BGL) has remained relatively constant since installation of the piezometer in the spring of 1983. The groundwater chemistry at this point is not similar to that of deep shale groundwaters sampled from other piezometers. The groundwater chemistry at J(s) (6 m.BGL) is represented by a series of numbers which indicate the changing major ion chemistry of the water since piezometer installation. The temporal change may be due to drilling water gradually being replaced by formation water in the vicinity of the piezometer intake. This long time period of mixing is not unusual considering the low hydraulic conductivity of this formation and more importantly the almost non-existent hydraulic gradient which drives groundwater flow.

The conclusion to this discussion on major ion chemistries is that only a few, near-surface, down-gradient piezometers (8, 3B, D(s), E(s)) may be monitored to detect changes in major ion chemistry caused by fly ash leachate contamination. However, since the vertical hydraulic gradient at most of these piezometers is strongly downward, it is doubtful whether contaminated groundwater originating from the ash landfill will migrate towards these piezometers. The groundwater chemistries determined from the remaining piezometers already have major ion chemistries similar to that of fly ash leachate being sampled from the sump, or the concentration of specific ions (Cl^-, TDS) are higher in the groundwater than in the leachate.

2.5.2 Minor and Trace Element Chemistry

Since the major ion concentrations in both fly ash leachate and bedrock groundwaters at Canada Brick are very similar, minor and trace element chemistry must be used for monitoring purposes. The most commonly used ions for such monitoring are boron and strontium. These elements can be used if their concentrations in ash leachate are significantly higher than in the natural groundwaters.

Figures 9a and 9b represent the changes in strontium concentrations over time in up-gradient and "on-site" piezometers respectively. The concentration of strontium found in the sump water is also plotted for comparison purposes. In most cases the strontium concentration in the sump water (represented by the inverted triangle) is similar or substantially below the concentrations in the up-gradient and "on-site" piezometers. The near-surface piezometers are filled with formation water representative of the weathered horizon.

Figure 10 represents strontium concentrations found in the down-gradient piezometers. The strontium levels in the sump water are similar to those in all the sampled piezometers. In conclusion, assuming that the sump water

correctly represents the in situ fly ash leachate chemistry, strontium does not appear to be a good indicator of fly ash contamination at this time. The number of plotted points are different between Figures 9 and 10 since sampling frequencies were also different as explained in Section 2.5.

Figures 11 and 12 represent changes in boron concentrations in up-gradient, "on-site" and down-gradient piezometers. For the most part, the boron concentration in the sump water is similar, or 1 to 2 mg/L higher than background values. The boron concentration in the sump water has been steadily increasing since about June 1985. No associated increases in boron concentrations are evident in the sampled piezometers. If the boron concentration in the sump water (and fly ash leachate) continues to increase, as the water level in the ash fill continues to rise, boron will become a more effective indicator of contamination by fly ash leachate.

Figure 13 represents the change in concentration of the trace elements, arsenic and selenium, in the sump water. The concentration levels tend to fluctuate wildly due to the periodic pumping of the sump. There is, however, a trend of increasing concentrations through time, with arsenic being more dominant than selenium. These trace elements will serve as effective indicators of ash leachate contamination since they are almost absent in the local groundwater. However, whether these trace elements will actually travel with the flowing groundwater, or be precipitated or absorbed onto the shale, remains a question.

At the present time, the sump water is not representative of fly ash leachate, especially in terms of minor elements such as strontium and boron. For comparison purposes, the most recent chemical analysis of fly ash leachate from the Birchwood Park site can be used since Lakeview TGS fly ash was also used for landfilling at this site. In July, 1988, the boron concentration in the fly ash leachate was 19 mg/L and the strontium concentration was 11 mg/L/4/. Comparing these values to the boron and strontium concentrations plotted for the sump in Figures 9 through 12, it may be concluded that the sump water chemistry does not, at this time, represent the true fly ash leachate chemistry at the Canada Brick quarry.

The pH recorded from the central sump fluctuated from 7.1 to 8.8 pH units but averaged 8.0 over the entire monitoring period. This average value is equivalent to that recorded in down-gradient wells BH-8 and BH-G and up-gradient well BH-5A/3/. By comparison, "in-ash" piezometers at the Birchwood Park site yield water samples with an average pH value of 9.4/5/, which is more representative of ash leachate.

The fly ash leachate chemistry from Birchwood Park is represented, in terms of major ion chemistry, in Figure 22. Comparing this plot with the Piper diagram of Figure 5, it can be seen that sulphate concentrations are more dominant in the fly ash at Birchwood Park than in the sump water at Canada Brick. This suggests that the sump water chemistry may in fact represent a mixing of deep bedrock groundwaters (BH-2B and 5B) with fly ash leachate. The sump may therefore not be a good sampling point for determining in situ leachate chemistry.

2.6 Conclusion

Fly ash landfilling began at the Canada Brick Quarry in December of 1981 and ended in December of 1986. Many piezometers were installed in the vicinity of the quarry during the initial site investigation and these piezometers have been monitored throughout the landfilling operation. Periodic water level measurements and water sampling have been used to determine if the ash landfilling has impacted on the local groundwater system.

Based on the periodic water level measurements, there has been no significant impact on groundwater flow characteristics under the residential subdivision down-gradient of the site. In the southeast corner of the Canada Brick Quarry property, some anomalous water level measurements indicate that there has been some reversal of hydraulic gradients immediately down-gradient of the site. Whether these reversals are related to the rise in water level within the ash landfill remains as yet, unresolved.

The determination of major, minor and trace element chemistries of waters sampled from down-gradient piezometers has not revealed the presence of fly ash leachate to date. This is not surprising since the water level within the fly ash landfill (as represented by the sump) has remained below water levels in the down-gradient piezometers (see Table II). Contaminant transport under this scenario would primarily be by the slow process of diffusion along the predominantly horizontal fracture network in the shale.

Because of the fracture transport dominated environment, the actual flow path of fly ash leachate away from the landfill could be quite complex. Therefore, it is impossible to know if the existing monitoring network is completely adequate for detecting future contaminant migration. A more successful monitoring system within the fractured shale would consist of a series of multilevel piezometers located along the down-gradient perimeter of the landfill site. These new monitoring points would have to be placed such as to completely intercept groundwater flow within the top 15 m of bedrock. Additional monitoring points within the shale immediately below the fly ash landfill would allow a more complete delineation of any potential contaminant plume.

Because of some uncertainty in the fly ash leachate chemistry determined from sampling the central sump, additional sampling points should be installed within the fly ash landfill. Water chemistries determined from these new points would be more representative of in situ fly ash leachate, and would therefore be more helpful for impact assessment.

3.0 BOOTH QUARRY FLY ASH LANDFILL

3.1 Background

3.1.1 Quarry Operations

This fly ash landfill site is located in the Borough of Etobicoke, southeast of the QEW-highway 427 intersection. The site was operated as a shale quarry from 1914 to 1974 by the Booth Brick Company and subsequently sold to the Borough of Etobicoke. The quarry was about 4.4 hectares in area with an average depth of 12.2 m (40 ft) below ground level.

3.1.2 Landfilling Operations

Reclamation of the quarry was achieved by landfilling with fly ash trucked from the nearby Lakeview Generating Station. The ash was placed in 8 to 12 inch layers and then compacted to 6 to 9 inches to achieve a target density of 1.6 Mg/m^3 at a moisture content of 25%/6/. This placement method provided for a disposal capacity of 816 Gg (900,000 tons) of ash. Landfilling operations began in 1976 and ended in 1979.

To facilitate drainage of the pit prior to filling, a ridge of lightweight aggregate was placed along the centreline of the quarry floor, ending in a depressed sump at the southeast corner of the quarry. The sump consisted of a 36-inch diameter, perforated culvert, positioned vertically and surrounded by gabion rock and crushed stone. The sump also provided an adequate water supply for spraying of the fly ash to control dusting during placement. It also allowed for monitoring and sampling of fly ash leachate after reclamation of the quarry.

3.1.3 Piezometer Installation

During the initial site investigation in 1975, 5 piezometers were installed along the southeast perimeter of the quarry/7/. All piezometers were seated 0.5 m into the shale bedrock and were used, along with the sump, for chemical sampling and groundwater monitoring during the filling operations. Between 1975 and 1982, several piezometers were destroyed and re-established such that since 1982, only the sump and four piezometers are being monitored and sampled regularly. These piezometers are located along a north-south section at the eastern boundary of the property, as shown in Figure 14. The piezometers installed in 1982 (OW1-82 and OW6-82) are located between 2.5 and 3.0 m into the shale bedrock (approximately 5.0 m.BGL).

3.2 Geology

The overburden in the area consists of a 2.4 to 3.0 m thick layer of stiff to very stiff mottled clayey sandy silt. An average soil hydraulic conductivity of 4×10^{-8} cm/s has been measured in the laboratory/7/. The shale bedrock is of the Georgian Bay formation, composed primarily of blue-grey shale with thin interbeds of limestone. This is the same shale which is exposed at the Canada Brick Quarry in Mississauga.

A transmissivity of 100 gpd/ft for the shale was estimated from observed groundwater leakage into the quarry/4/. Following landfilling with fly ash, the water table in the ash was expected to rise to the level of the regional water table within 10 to 15 years.

3.3 Water Level Monitoring

Water level elevations have been recorded in five piezometers and the central sump, since the spring of 1982 when the last set of piezometers was re-established/8/. These data are plotted in Figure 15. Piezometer OW7-82, installed in soil fill material in the southwest corner of the quarry, shows a water level rise similar to that of the central sump. These two data sets represent the rate of rise of water within the quarry during a four-year period equal to approximately 0.8 m/yr.

The remaining piezometers represent water levels in the surrounding bedrock and overburden and are influenced by local horizontal and vertical hydraulic gradients. Up until August 1986, the water level in the quarry was still below the water levels in the surrounding piezometers. This indicates that the flow of groundwater was still predominantly towards the quarry ten years after landfilling operations began. During this period, any leachate migration from the quarry would have been diffusion-controlled.

At the rate of water level rise indicated in Figure 15, levels within the quarry should have approached down-gradient levels in mid-1988. However, because of limited access to the original sump location and to the surrounding piezometers, more up-to-date water level measurements are unavailable.

From past water level measurements, the regional water table in the vicinity of the Booth Quarry is within 1 to 1.5 m below ground surface and slopes southward towards Lake Ontario under a horizontal gradient of 0.005/6/.

3.4 Geochemical Monitoring

The major ion chemistries of waters sampled from the Booth Quarry piezometers and sump are shown on the Piper diagram of Figure 16. The water chemistries from the four piezometers are represented by one point each because the ion concentrations have remained relatively constant throughout the sampling period. The full set of chemistry results is tabulated in other reports/8/.

Piezometers OW-1-82 and OW-6-82 were installed 5 m below ground surface which is within the weathered zone of the bedrock. Piezometers OW-2-77 and OW-8-77 were installed at the overburden/bedrock contact or 2.5 to 3 m below ground surface. The water chemistries from all four piezometers are similar to those from the near-surface piezometers at the Canada Brick Quarry where the weak acid anion concentrations (HCO_3^-) are similar or higher than the strong acid anions ($SO_4^=$, Cl^-). To date, there is no evidence of a shift in major ion chemistry of groundwater in the near-surface zone.

The major ion chemistry of the sump water is plotted as a series of numbers representing sampling dates, ie 1 = Oct, 1977 and 16 = Aug, 1986. Until 1979 (point 6), the sump water was being pumped for spraying the landfilled ash to control dust. Up until June, 1982 (point 10) the sump water appeared

representative of a mixing of deep shale groundwaters with fly ash leachate. In late 1982, the water chemistries indicated that a massive dilution of all ion concentrations occurred. In addition, bicarbonate ions became more dominant than sulphate ions in the sump water. It is believed that this phenomenon was a result of leakage of surface water into the sump following site redevelopment.

Because of the questionable water chemistry results obtained from the sump, it is impossible to know what the present concentrations of fly ash indicators (boron, strontium, arsenic and selenium) are within the landfilled quarry. From the piezometers in the vicinity of the quarry, the background concentration of boron ranges from 0.2 to 3.4 mg/L and that of strontium ranges from 0.3 to 5 mg/L. The concentrations of trace elements arsenic and selenium are below detection. No consistent increase in concentrations of the above elements has been observed in the down-gradient piezometers.

3.5 Conclusions

Based on measurements from the sump within the Booth Quarry, the water level within the landfilled ash probably reached background water levels in mid-1988. Before this date, local groundwater flow has been predominantly towards the quarry, preventing any substantial migration of ash leachate away from the site. To date, no consistent increase in fly ash indicator elements (ex. boron and arsenic) has been detected in the down-gradient monitoring points. However, since the monitoring network consists of only four individual piezometers placed in the near-surface zone of the shale, it is doubtful whether any evidence of contaminant migration will ever be found. Groundwater flow in this type of geological environment will be predominantly through the horizontal fractures in the shale. Unfortunately, no data exist on the variation of hydraulic conductivity with depth or on the existence of preferential flow paths within the top 14 m of bedrock in contact with the landfilled fly ash. This entire thickness of bedrock should be tested and monitored for fly ash leachate especially since the landfill has just recently come into hydraulic equilibrium with the surrounding bedrock.

Monitoring the impact of ash leachate migration into the surrounding groundwater system would be greatly improved by: 1) installing one or more monitoring points within the ash fill to determine source terms and water levels, and 2) installing multilevel monitoring devices immediately down-gradient of the quarry to allow hydraulic conductivity determinations and water sampling along the entire section of shale bedrock in contact with the ash. This latter activity could prove to be very difficult to accomplish since the area immediately south of the quarry has been completely developed for industrial purposes, making site access almost impossible.

4.0 BIRCHWOOD PARK - FLY ASH LANDFILL SITE

4.1 Background

Birchwood Park (formerly known as Clarkson Fly Ash Park) is located on the north side of Lakeshore Rd (Hwy #2) in Clarkson, Mississauga, southeast of the QEW-Erin Mills Parkway interchange (Figure 17). The park area covers

approximately 8.6 hectares, and was operated as a sand pit before being obtained by Ontario Hydro for landfilling purposes. The bottom of the pit was 6.4 m below ground surface at its deepest point.

Reclamation of this site, using fly ash from the Lakeview Thermal Generating Station, occurred during the period between 1968 and 1972. The filling sequence was in four phases, progressing west to east and finally south. The fly ash depth reaches a maximum of 17 m under the peak of the "scenic mound", with 10 to 15 m depths being common under the bulk of the ash pile. The contractor was allowed to place the fly ash in 1.5 m (5 ft) thick lifts which caused the fill to vary from very loose (bulk density = 1470 kg/m^3) to very dense (2100 kg/m^3) in a random fashion/9/.

Because of a scarcity of published field data concerning the migration of contaminants from fly ash landfill sites, a detailed groundwater study was undertaken in 1978-1980 at the Birchwood Park site. Not only was this site close to Ontario Hydro's research laboratory, but the fly ash had been in place in the abandoned pit for roughly 10 years. It was believed that sufficient time had elapsed for potential contaminant plumes to have fully developed in the groundwater system beneath the landfill. An additional field study was conducted in 1984-1985. A total of 47 piezometers were installed at the site and were used for hydraulic head monitoring, hydraulic conductivity testing and groundwater sampling (Figure 17). Piezometers were located primarily in the direction of groundwater flow with more piezometers at the down-gradient side of the site to help detect the advance of potential contaminant plumes. The locations of the piezometer tips are shown along three cross-sections in Figure 18.

4.2 Site Stratigraphy

The site stratigraphy, as determined by drilling and sampling is shown along the main cross-section in Figure 19. Underlying the fly ash is a silt layer having a maximum thickness of 5 m. The hydraulic conductivity (K) of this layer, as determined by laboratory and field tests (falling head tests), ranges from $<1 \times 10^{-6}$ to 5×10^{-4} cm/s. Toward the southeast side of the filled area, a sandy silt stratum of variable thickness lies between the ash fill and the silt layer. K values range from 4×10^{-5} to 3×10^{-4} cm/s for this sandy silt layer. Underlying the silt is a 3-4 m thick layer of clayey silt with K values ranging from $< 1 \times 10^{-7}$ to 1×10^{-6} cm/s. This clayey silt is underlain by a 1-2 m thick layer of sandy silt till ($K = 1 \times 10^{-6}$ cm/s) which directly overlies the Dundas shale bedrock, common to the entire Metropolitan Toronto area.

4.3 Groundwater Movement

The only continuous set of water level measurements is for the period 1978-1979. Two additional sets were taken in the summer of 1985/10/. These measurements indicate that the water table is located very deep within the fly ash pile. At borehole 4, the water table is 13.3 m below the new ground surface which means that a substantial portion of the fly ash pile is in an unsaturated condition. In this case, hydraulic conductivity of the ash pile will be several orders of magnitude less than the saturated conductivity of 1×10^{-5} cm/s/9/.

Groundwater flows into the site from the west corner and traverses the site in a southeasterly direction, in general from boreholes 5 to 14/9/. The calculated horizontal hydraulic gradient between boreholes 4 and 1 is approximately 0.02. Some groundwater also flows easterly between boreholes 6 and 8 under a hydraulic gradient of 0.01. Insufficient data is available for estimating vertical hydraulic gradients. However, since the surficial materials are very thin relative to their horizontal extent and the site is not far from the shoreline of Lake Ontario, it is reasonable to assume that the groundwater flow is predominantly horizontal.

Based on potentiometric data for August 30-31, 1979, a potentiometric surface, or water table in this case, was generated for the Birchwood Park site. This surface, shown in Figure 20, was generated by kriging, which is a statistical method for estimating the value of a function at a point, given spatially-distributed observations of the function at a number of points/8/. The kriged phreatic surface was used to calibrate two-dimensional and three-dimensional flow models of the site for contaminant migration modelling/10/. Aside from the hydraulic head values themselves, additional data representing the level of two bounding creeks were input to the kriging program. These creeks are located on the northern and southern boundaries of the site and were thought to play an important role in the contaminant migration process. The conclusions of the kriging analysis and the multidimensional flow modelling were that: 1) groundwater flow is predominantly horizontal, 2) there is considerable underflow at both the northern and southern creeks, 3) most of the flow occurs through the more permeable fly ash zones below the water table and through the sandy silt materials adjacent to the fly ash.

Although the initial estimates of groundwater flow directions have been substantiated by the modelling effort, Figure 20 also indicates that a portion of the groundwater flow is towards the northeast where no piezometers have been installed.

4.4 Groundwater Geochemistry

The results of groundwater sampling and chemical analysis are available for the periods August. 1978 - June, 1980 and January, 1985 to September, 1985. Although up to 34 chemical constituents were analyzed for in the groundwater, the study of sulphate concentrations was sufficient to determine the extent of groundwater contamination below the ash fill. The choice of sulphate was made for the following reasons: 1) sulphate is a major constituent of fly ash leachate, 2) the background concentration of sulphate in the groundwater is substantially less than in the leachate, 3) sulphate is least susceptible to sampling and analytical errors, 4) sulphate was not expected to be readily attenuated or retarded by the underlying sediments/10/.

Examination of the sulphate concentration cross-section shown in Figure 21 reveals that the plume of elevated concentrations beneath the landfill site has become thinner and has migrated approximately 50 metres down-gradient. Only the 1000 mg/L contour has been plotted for the sake of simplicity. However, contour lines up to 2500 mg/L are shown on the original plots/10/. Examination of sulphate concentrations with time, for piezometers located directly underneath the ash pile (BH 4, 12 and 2) reveal an apparent trend of

decreasing concentrations with time. Similar behaviour was also noted for other leachate constituents such as Na^+ and Ca^{2+}. The piezometers located on the down-gradient side of the fill (BH-22, 24 and 27) showed a general increase in concentration with time for all three constituents/10/. The Piper plots shown in Figure 22 confirm the gradual change in groundwater chemistry in the down-gradient piezometers, 1C and 22, from that of background water chemistry to that more representative of fly ash leachate.

The results of the groundwater monitoring program at Birchwood Park provided the following conclusions/10/:

1. The major fly ash leachate components (calcium, sodium and sulphate) showed elevated concentrations below and down-gradient from the ash fill. For the case of sulphate, values ten times above background have been recorded, ie 2000-2500 mg/L.

2. Minor leachate components such as potassium, boron, aluminum and strontium exhibited a small zone of elevated concentrations beneath and down-gradient from the fill. For the case of boron, values between 10-15 mg/L have been recorded which can be up to ten times above background.

3. It was unclear whether trace leachate components such as chromium, selenium and lead exhibited limited zones of elevated concentrations, due to the incompleteness of the data.

4. The apparent trend of decreasing sulphate concentration with time underneath the ash fill may be related to: i) higher precipitation/infiltration at the start of the monitoring period, or to ii) a passage of a first flush of leachate released during filling and prior to capping of the fly ash.

4.5 Conclusions

A poor initial landfill design, ie no clay liner or leachate collection system, and subsequent poor landfilling practices have resulted in a plume of major ions such as sulphate, calcium and sodium, to develop down-gradient of the ash fill, but still within the park boundaries. Migration of fly ash leachate appears to be concentrating in the upper sandy silt horizon which exhibits a higher hydraulic conductivity than the underlying silt unit.

Fly ash leachate is being primarily generated by groundwater flowing through the saturated fly ash below the water table. Computer simulations of the movement of the contamination plumes at this site indicate that groundwater contamination could have been considerably reduced if the ash was located completely above the water table/10/.

Infiltration of surface water would be minimal because of the thick unsaturated portion of fly ash which has a low hydraulic conductivity. Thus, fly ash leachate generation would have been slow. It may be concluded that an improved ash fill design (ie ash placement, liner thickness, etc) aided by a computer model, could have minimized the extent of groundwater contamination.

The possibility also exists that added precautions during the filling operations could have reduced the contamination at the site. The filling operation spanned several years during which time the ash was exposed to precipitation. In addition, the ash was periodically sprayed by a water truck to avoid dusting problems. During this time, it is conceivable that significant volumes of leachate could have developed which would eventually infiltrate into the natural groundwater regime. It is possible that had precautions been taken to cover the ash so that exposure to the elements during filling was reduced, the contamination at the site could have been reduced.

It remains difficult at this time to determine the full impact of fly ash landfilling on the surrounding groundwater system. No data has been collected on any past and/or present users of groundwater from the near surface sediments down-gradient of the site. No piezometers have been installed northeast of the ash pile even though numerical modelling results indicate the possibility of groundwater flow in that direction. Several piezometer tips are located in the deeper, lower permeability horizons especially in the northern and southern portions of the site. Since the near surface sandy silt horizon has a higher permeability, it would seem to be a preferential flow path for any fly ash leachate. The geochemical data base for this site is sparse and incomplete which tends to make any impact assessment rather difficult. The importance of the creek as a local discharge zone along the southern boundary of Birchwood Park has not been determined in the field.

5.0 FINAL CONCLUSIONS

The hydraulic and geochemical data have been reviewed for the following fly ash disposal sites: Canada Brick Quarry, Booth Quarry and Birchwood Park. The principal objective of this review was to determine if fly ash landfilling had impacted on the local groundwater regimes. At the Canada Brick and Booth Quarry sites, an impact assessment based on geochemistry is complicated by the fact that the local groundwater chemistry, derived from the shale bedrock, is very similar to fly ash leachate chemistry. Any contamination study must be restricted to examining minor (boron and strontium) and/or trace (arsenic and selenium) element concentrations.

Based on the hydraulic and geochemical data available to date, the following comments can be made:

1. No evidence of fly ash leachate contamination has been detected at the Canada Brick site to date. Since this disposal site has only recently been completed, it has not yet come into hydraulic equilibrium with the local groundwater regime. Monitoring should continue within the ash fill and selected down-gradient piezometers. The installation of additional monitoring points within and underneath the ash fill and of multilevel piezometers immediately down-gradient of the site is recommended. This would greatly enhance the quality of data being collected and increase the validity of future impact assessments.

2. No evidence of fly ash leachate contamination has been found at the Booth Quarry since landfilling ended in 1979. Based on water level readings within the ash fill, the Booth Quarry has only recently come

into equilibrium with the local groundwater regime and therefore, regular monitoring of groundwater chemistry should continue into the future. The installation of additional monitoring points at this site, is recommended to allow for a more realistic impact assessment.

3. Elevated concentrations of major ions (Ca^{2++}, Na^{+}, SO_4^{2-}) have been detected in the unconsolidated sediments underneath and down-gradient of the Birchwood Park fly ash landfill. The migration of these elements can be attributed to poor landfill design and practice at this site. Periodic monitoring of groundwater chemistry, especially of minor and trace element concentrations, would enhance any future impact assessment.

Detailed recommendations for the design of fly ash landfills for the purpose of minimizing environmental impact have been provided by Chan, Ash and Sykes (1986)/10/. Some of the highlights of these recommendations are:

1. Based on data collected from a detailed site investigation and a knowledge of ash and ash leachate properties, the design of the landfill should evaluate whether or not clay liner or leachate collection systems are required.

2. The detailed site investigation should provide data on site lithology/stratigraphy, groundwater flow patterns and hydraulic conductivities, climatology and availability of construction materials.

3. The ash landfill design should be analyzed using a computer model which addresses questions such as infiltration to the ash, groundwater mounding and the development of contaminant.

4. Ash placement should be carried out entirely above the water table so as to maintain the ash in an unsaturated condition.

5. Infiltration to the ash fill should be controlled through the provision of an engineered cover.

6. The ash fill should be compacted so that excessive differential settlements will not occur which could damage the clay barrier.

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TABLE I

Piezometer Locations

Piezometer Number	Ground Elevation (m.ASL)	Elevation of Intake Bottom (m.ASL)
	137.5	122.5
*2B (D)	138.5	123.3
*2A (S)	138.5	130.9
3B (S)	120.6	110.4
3A (I)	120.8	106.6
3C (D)	120.7	96.3
*4	132.5	117.3
5B (D)	141.5	127.8
5A (S)	141.5	133.9
*6	142.1	132.9
*7	145.8	137.0
8	116.0	105.0
9	128.2	112.3
10A (D)	119.1	103.9
10B (S)	119.1	113.0
A	130.0	117.8
* Bs	121.9	121.0
* Bd	121.9	118.8
Cs	119.1	115.4
Cd	119.1	109.1
Ds	117.6	113.6
Dd	117.6	107.6
*Es	111.4	107.4
*Ei	111.4	101.6
*Ed1	111.4	90.2
*Ed2	111.4	87.0
Fs	108.1	103.2
Fd	108.0	97.9
G	116.2	106.9
*Hs	114.2	104.3
*Hd	114.2	104.3
*Is	110.9	103.3
*Id	110.9	100.8
Js	110.0	103.8
Jd	110.0	99.8
Es85	112.1	106.2
Ed85	112.1	88.1
Sump		

S = shallow piezometer
F = intermediate piezometers
D = deep piezometer

TABLE II

Canada Brick Ground Water Levels*
Sept 1982 - Sept 1988/3/

DATE/ LOCATION	ELEVATION (m) OF TOP OF STAND PIPE	82/09/20	82/11/03	83/02/08	83/03/01	83/05/26	83/09/12	84/01/08	84/04/02	84/06/08	84/09/30	85/03/14	85/06/01	85/09/12	85/12/10	86/03/25	86/06/10	86/10/06	86/12/17	87/03/16	87/06/08	87/09/.0	88/05/06	88/09/02	
1	138.4	124.46	124.56	124.49	124.50	124.47	124.45	124.50	124.61	124.57	124.54	124.60	124.56	124.51	124.58	124.51	124.55	124.57	124.4	124.35	124.43	124.38	-	-	
2A(s)	139.3	132.89	132.89	132.91	138.31	138.40	135.73	138.30	138.40	137.98	132.79	138.13	137.63	137.54	137.85	138.08	138.00	138.05	138.12	137.5	135.82	N/A	-	-	
2B(d)	139.2	127.86	127.61	128.13	128.25	128.07	127.90	127.73	127.90	127.90	128.45	129.10	129.40	127.25	128.01	128.04	127.96	127.46	127.92	130.12	129.61	128.93	-	-	
3A(i)	121.2	109.17	109.61	109.55	109.42	109.40	109.20	109.63	109.84	109.50	109.13	110.75	109.38	109.71	109.72	111.57	111.96	112.90	112.8	112.2	111.73	112.18	110.76	110.76	
3B(s)	120.8	113.69	113.80	114.01	114.03	113.96	113.30	113.68	114.22	114.10	113.10	114.90	111.55	114.70	116.69	116.56	113.88	115.83	114.78	113.89	114.42	113.42	114.87	112.38	
3C(d)	121.2	107.84	108.00	108.35	108.07	111.55	111.50	111.52	111.77	111.55	111.47	111.86	111.56	111.58	113.58	112.08	113.17	114.2	113.42	112.87	113.06	116.44	111.91	111.46	
4	133.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
5A(s)	141.8	-	136.55	136.65	136.80	136.82	136.18	136.05	141.11	140.15	136.03	140.10	140.10	139.85	136.61	141.12	141.08	-	-	430.95	138.7	136.36	135.93	134.91	
5B(d)	141.8	-	131.66	132.02	032.22	132.15	131.51	131.88	131.83	131.36	131.01	131.55	131.36	131.12	131.12	131.32	131.40	130.94	131.02	136.7	131.1	134.74	131.43	130.15	
6	142.6	-	-	-	-	135.85	135.17	125.25	135.77	135.55	135.12	135.75	135.34	135.19	135.53	135.70	135.42	135.72	135.71	135.63	135.38	-	-	-	
7	146.3	-	-	-	-	143.40	142.39	142.75	143.41	143.17	142.41	143.55	142.73	142.52	143.23	143.97	143.62	143.98	143.88	143.91	143.19	-	-	-	
8	116.5	109.81	109.81	109.87	110.05	110.07	109.70	109.86	110.09	110.02	110.25	110.65	109.85	109.91	110.27	110.91	111.88	113.15	112.86	112.73	112.68	-	112.35	110.47	
9	128.7	113.12	113.12	112.60	112.59	112.60	112.61	-	122.25	121.80	121.64	122.09	121.79	121.95	121.97	122.06	122.06	124.89	126.29	121.95	121.59	121.7	-	-	
10A(d)	119.5	111.97	111.97	112.19	112.18	112.40	112.21	-	112.48	112.22	-	112.37	112.27	112.15	112.09	112.10	-	112.31	112.28	112.25	112.14	112.13	112.31	111.5	
10B(s)	119.4	116.93	116.93	117.56	117.56	117.69	116.73	-	117.65	117.28	-	117.74	117.03	117.35	117.65	117.60	-	117.88	117.81	117.66	N/A	119.4	116.56	115.51	
A	130.4	121.74	121.74	-	121.91	121.59	121.78	121.80	121.88	121.84	121.69	121.75	121.65	121.65	121.66	121.53	121.49	121.59	121.69	121.36	121.35	121.32	121.12	120.00	
B1	122.0	Permanently damaged during local construction						-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
B2	122.4							-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
C1	119.0	112.17	112.00	112.00	111.84	111.65	112.10	111.88	111.83	111.53	112.00	111.60	111.50	112.19	112.30	111.83	111.75	112.69	112.48	111.81	111.64	112.25	111.47	111.08	
C2	-	116.34	116.49	117.82	117.72	117.84	116.22	116.51	117.99	117.07	116.19	118.11	116.42	116.96	118.00	117.01	116.74	118.00	118	117.63	116.2	116.05	117.37	115.10	
D1	110.09	111.05	-	110.50	110.34	110.25	109.98	-	110.20	110.13	110.05	110.23	110.03	110.20	110.30	110.25	110.20	110.50	110.4	110.26	110.11	110.25	110.06	109.12	
D2	116.12	115.46	115.80	116.20	116.18	116.08	115.20	-	116.20	115.74	115.09	116.22	1115.42	115.69	115.86	116.15	115.60	116.09	116.07	116.05	115.22	109.8	115.59	113.77	
E4-2	105.98	105.86	105.89	105.93	105.88	105.90	105.76	105.85	105.96	105.87	105.82	Damaged during local construction (Being Replaced)		ED85	106.17	106.15	106.18	106.42	106.81	106.73	106.21	106.99	106.17	105.19	
E1	-	106.36	106.45	106.30	106.61	106.60	106.38	-	106.78	105.33	106.38			-	-	-	-	-	-	-	-	-	-	-	
E2	-	108.53	108.60	108.69	108.78	108.87	108.34	108.85	108.73	108.74	108.64			E285	108.25	108.35	108.29	108.51	108.44	108.24	108.18	108.35	108.21	104.95	
F4	105.20	105.43	105.51	105.73	105.65	105.68	105.47	-	105.60	105.46	105.43	105.67	105.42	105.55	105.70	105.56	106.46	105.70	105.55	105.46	105.3	105.19	105.23	104.24	
F2	100.34	105.05	105.59	104.19	105.13	105.24	105.10	108.59	105.21	105.51	105.12	105.55	105.14	105.20	105.45	105.35	105.35	105.48	105.55	105.34	105.3	105.25	105.20	104.20	
G	110.25	109.97	109.95	110.10	110.10	110.16	109.87	105.47	110.10	110.06	109.36	110.63	110.23	110.13	110.10	110.93	110.94	111.55	111.21	110.97	110.6	110.45	110.13	109.59	
M1	-							105.10	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
M2	-							110.02	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
M3	-	Destroyed during local construction							-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
I1	-							-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
I2	-							-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
J1	-	-	103.83	105.70	105.76	105.70	105.70	-	106.15	105.31	105.54	106.26	105.66	104.96	105.62	105.83	105.59	105.28	106	105.99	105.77	105.87	105.77	104.69	
J2	-	-	106.32	106.05	105.95	105.92	106.13	-	106.00	105.87	105.86	106.04	105.68	105.80	105.93	105.83	105.90	106.18	106.1	106.07	105.82	105.88	105.73	104.80	
Sump	107.40	105.85	106.27	106.70	106.63	106.05	N/A	107.34	107.61	103.98	104.07	109.13	105.37	106.20	104.82	110.35	111.59	113.22	113.12	113.05	111.79	**	105.6**	110.58**	

* Suffixes d, i, s indicate depth of piezometers, to, deep, intermediate and shallow, respectively, for multilevel nests.
** Sump being discharged
\- Means level not available due to damage or inaccessibility
All level measurements are expressed in metres above geodetic datum

(C1)

LEGEND:

- Piezometer Location
- Canada Brick Property Boundary
- Cross-Section Location

SCALE:

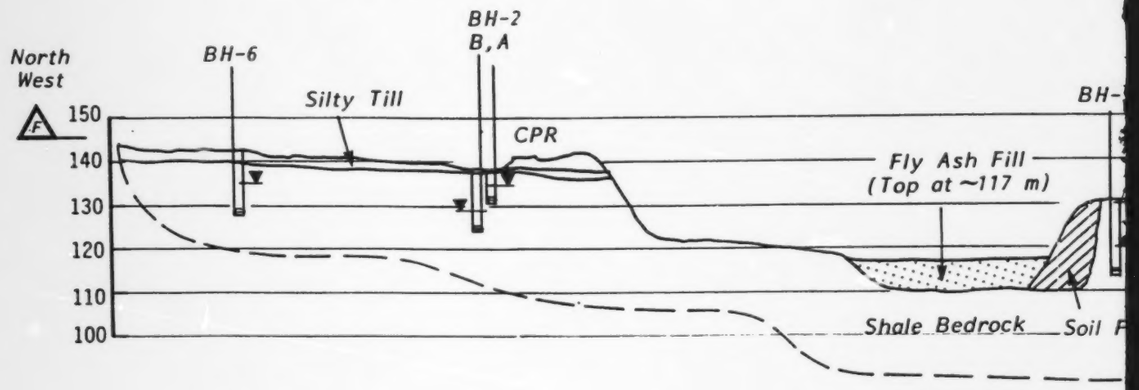
0 100 m 200 m

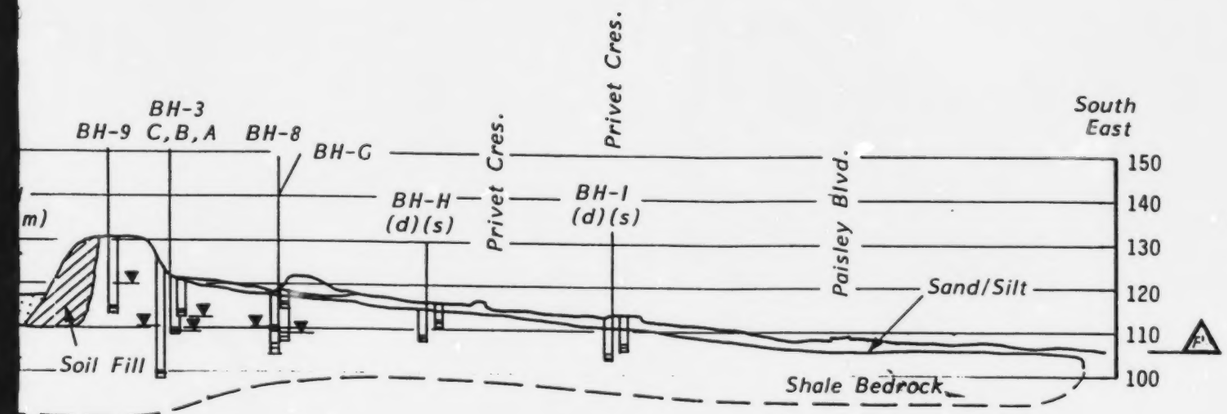
FIGURE 1

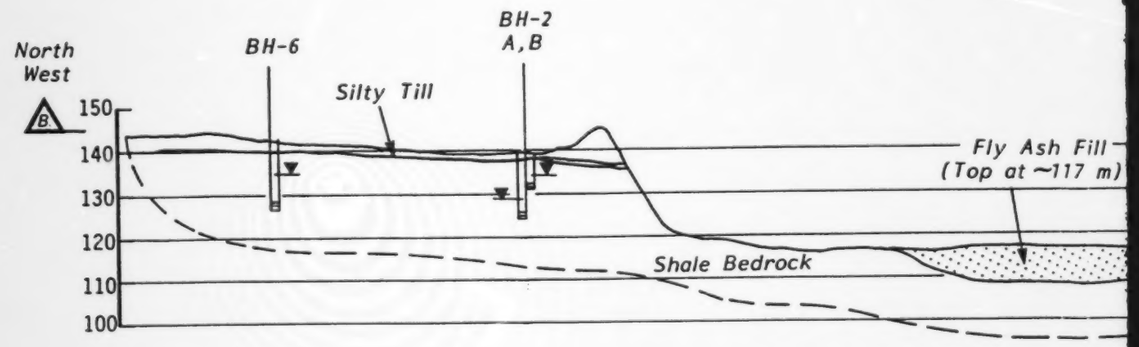
CANADA BRICK QUARRY
FLY ASH DISPOSAL PROJECT

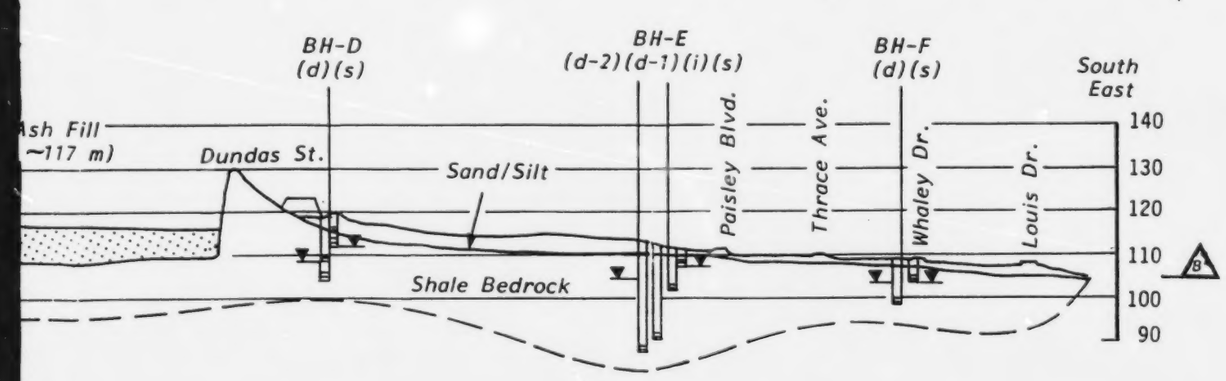
200977-RD

89-42









Horizontal Scale: 1 cm = 48 m

▼ = Water Levels (May, 1988)

FIGURE

CANADA BRICK
FLY ASH DISPOSAL

- Piezometer Locati
Two Cross-Se

FIGURE 2

BRICK QUARRY
ISPOSAL PROJECT

r Locations Along
ross-Sections

200978-RD

89-43

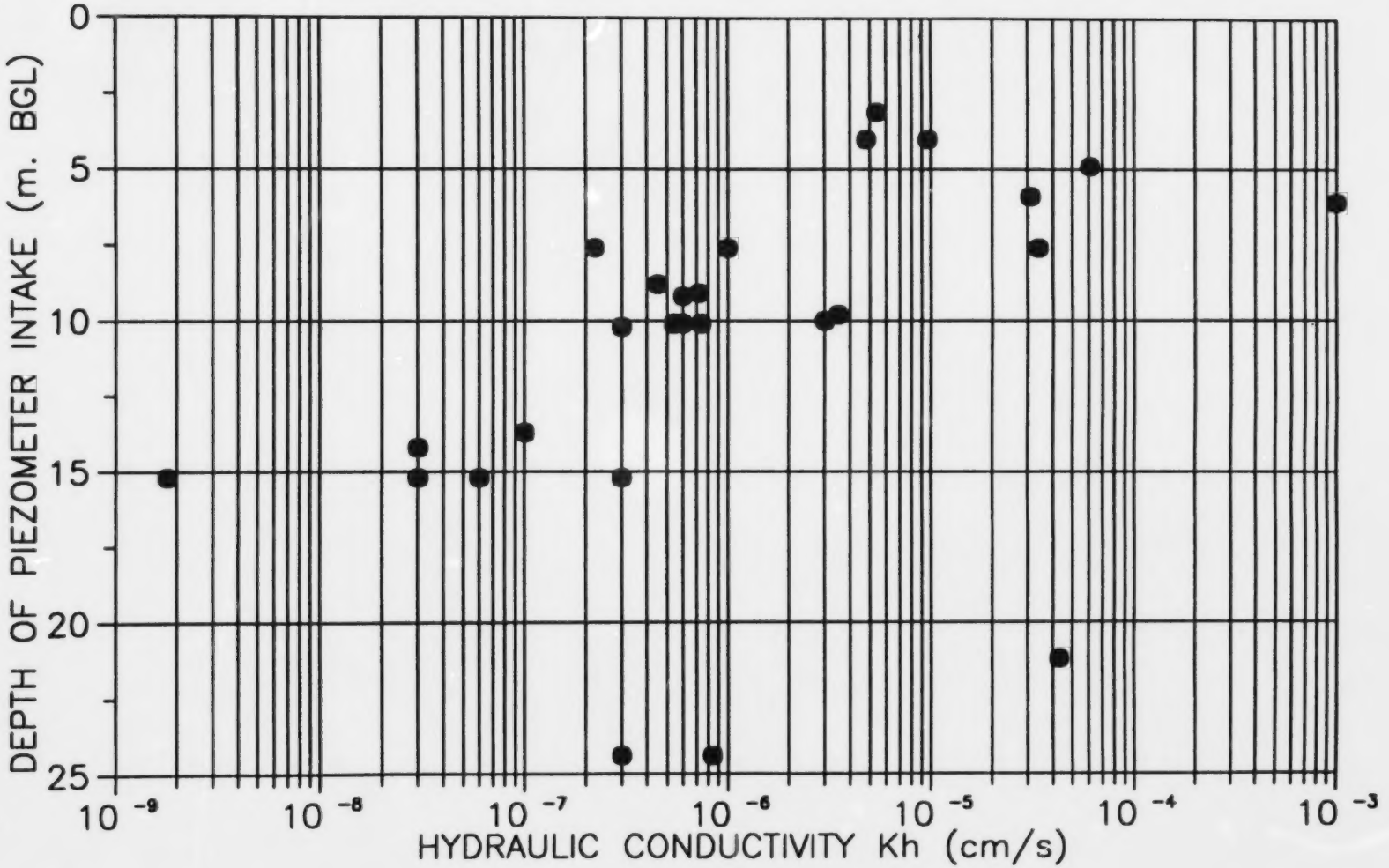
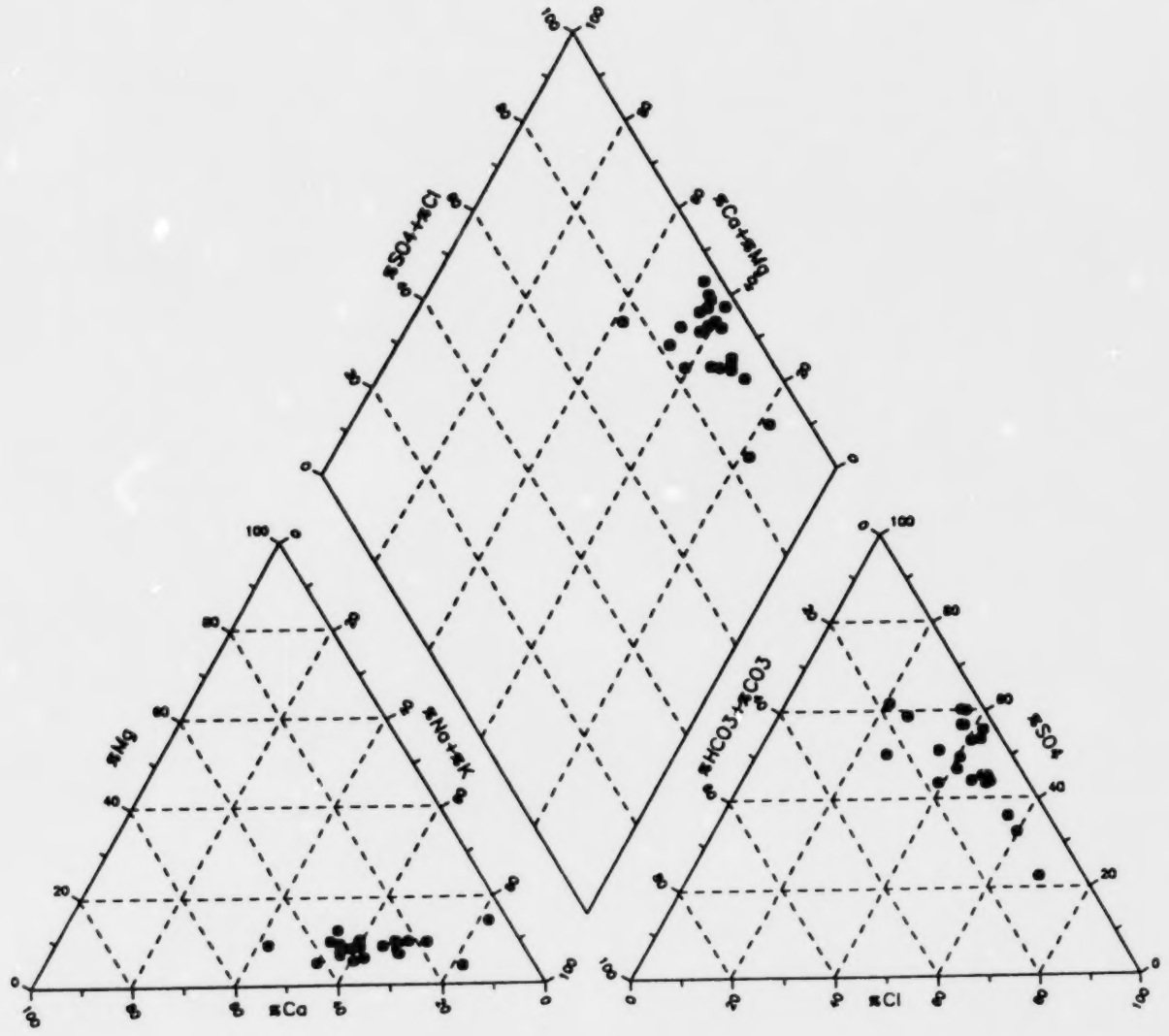


FIGURE 3: CANADA BRICK QUARRY
K_h vs. depth relationship for shale bedrock

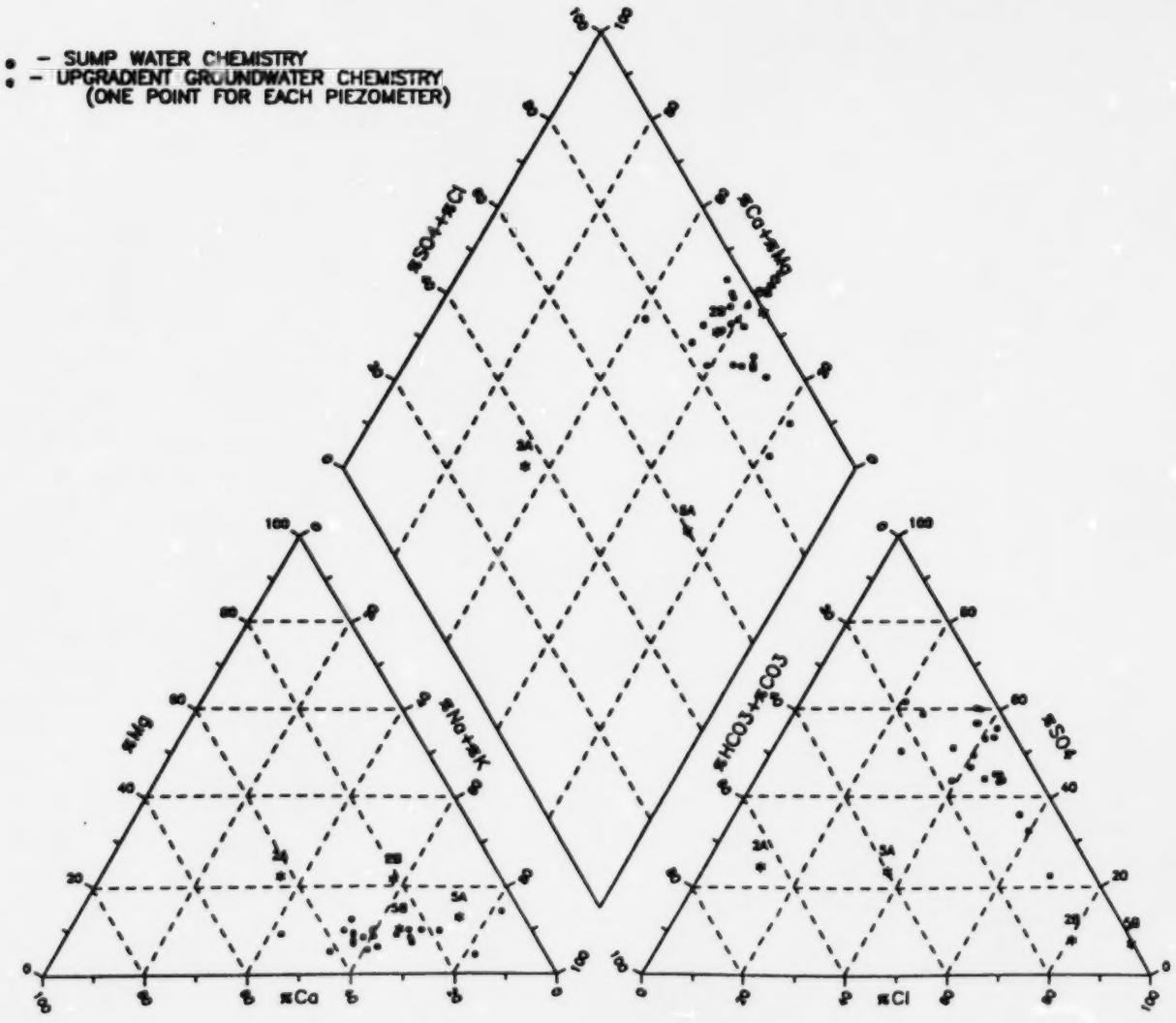
PIPER DIAGRAM
MAJOR ION CHEMISTRY

Figure 4 :
CANADA BRICK FLYASH DISPOSAL SITE
– Sump #1 water chemistry from
March/82 to June/87



PIPER DIAGRAM
MAJOR ION CHEMISTRY

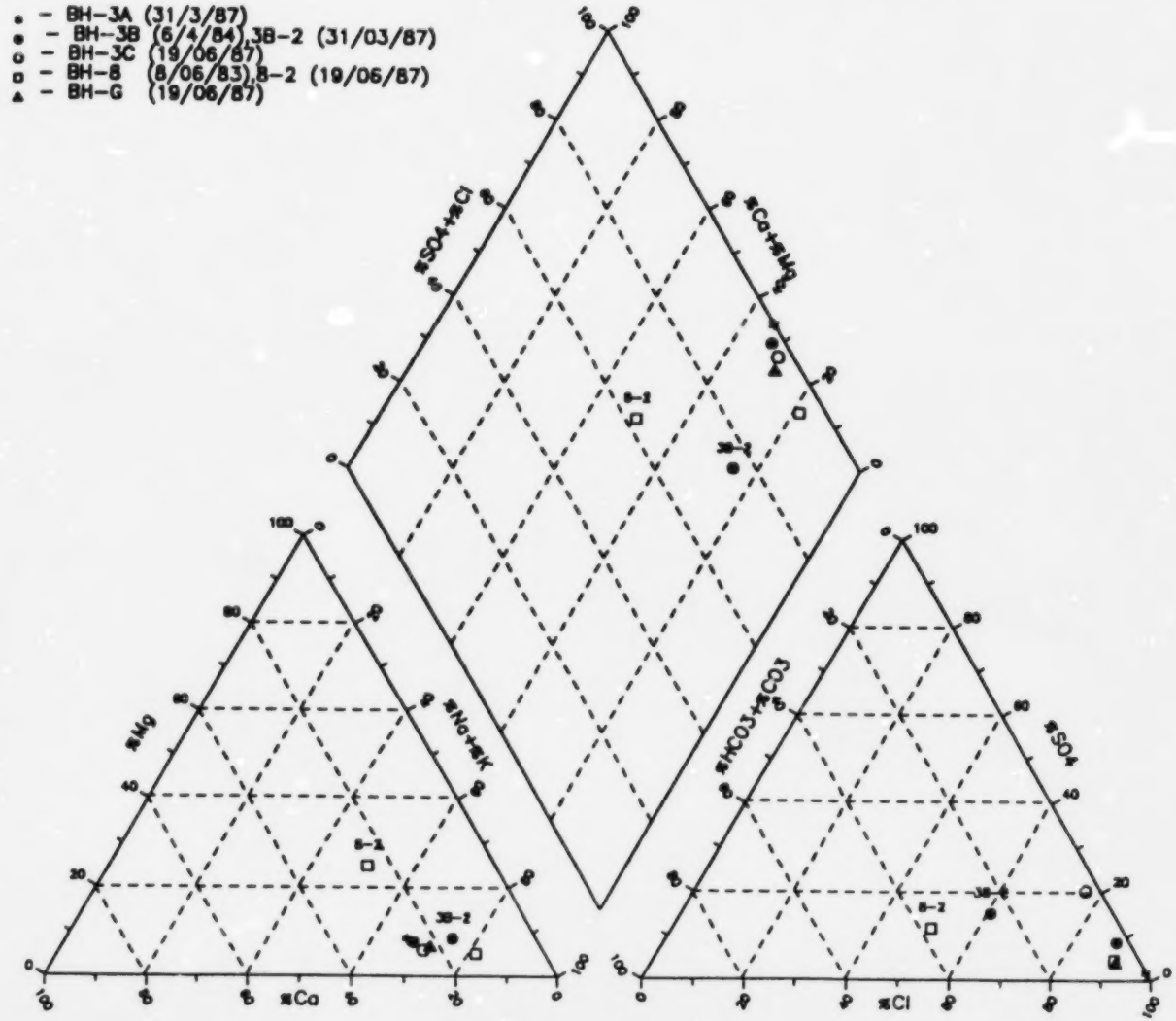
Figure 5 :
CANADA BRICK FLYASH DISPOSAL SITE
– Sump #1 water chemistry and upgradient groundwater chemistry



PIPER DIAGRAM
MAJOR ION CHEMISTRY

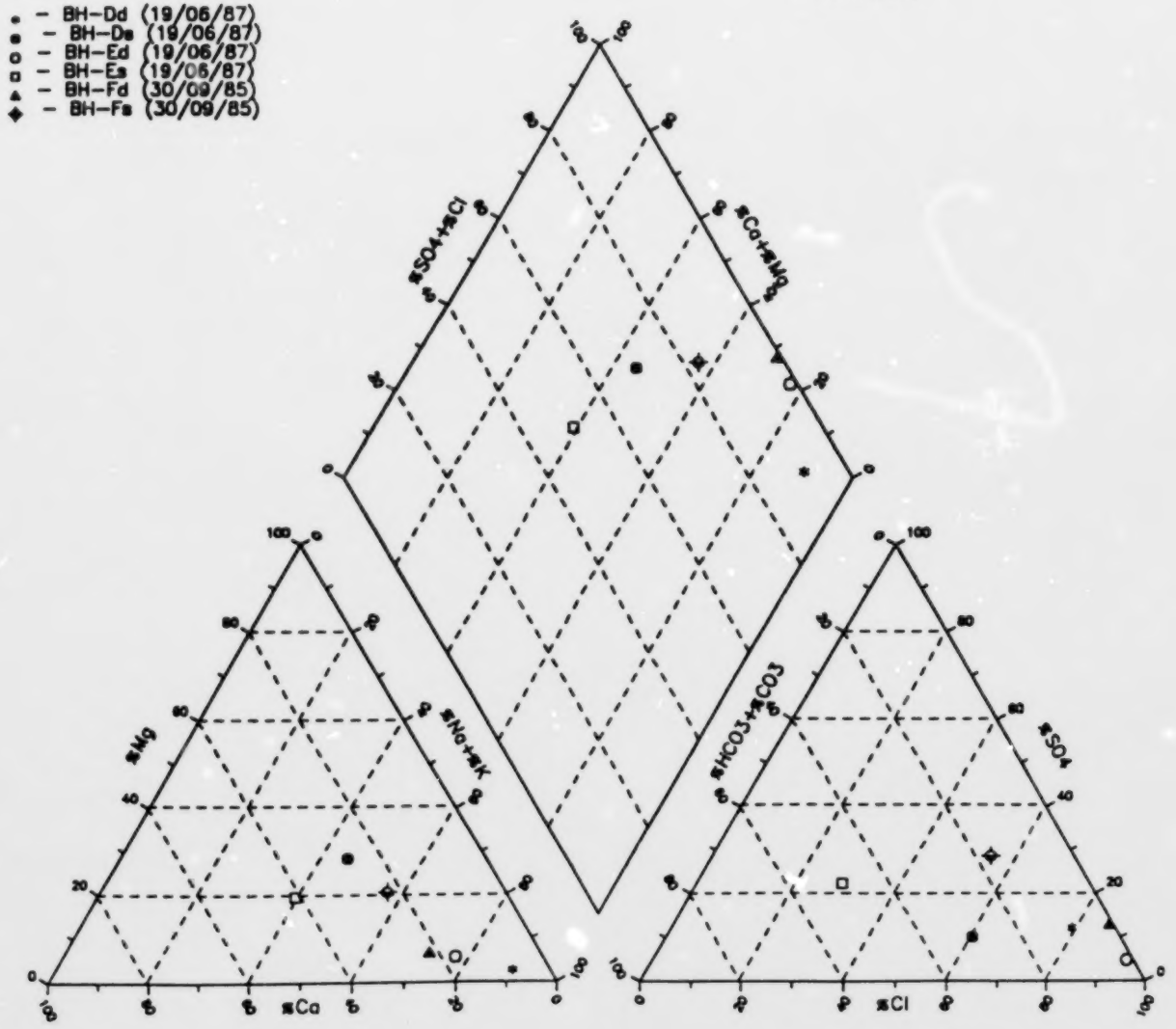
Figure 6 :

CANADA BRICK FLYASH DISPOSAL SITE
– On–site groundwater geochemistry downgradient of fly ash cells



PIPER DIAGRAM
MAJOR ION CHEMISTRY

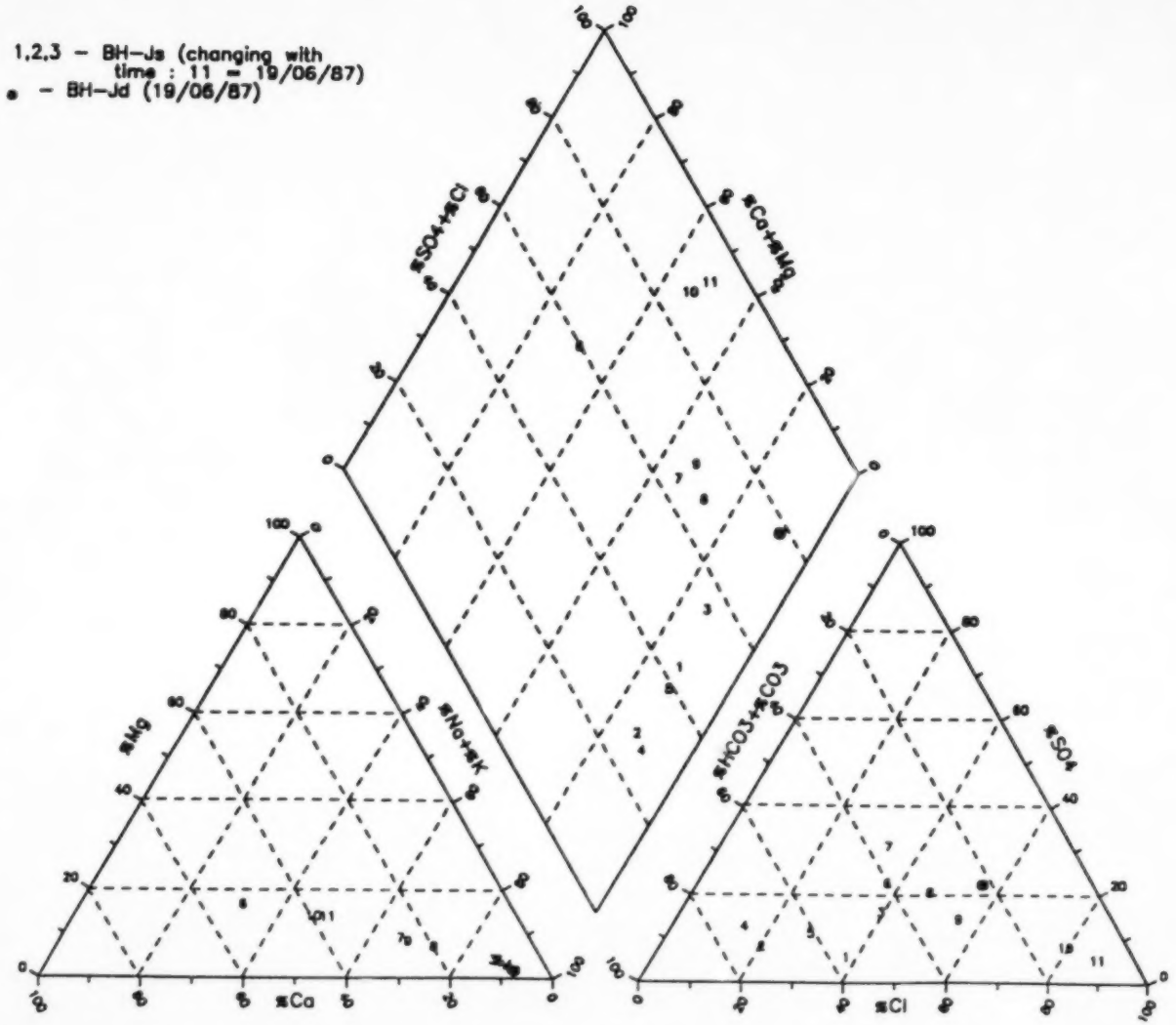
Figure 7 :
CANADA BRICK FLYASH DISPOSAL SITE
– Downgradient groundwater chemistry



PIPER DIAGRAM
MAJOR ION CHEMISTRY

Figure 8 :

CANADA BRICK FLYASH DISPOSAL SITE
– Downgradient groundwater chemistry



CANADA BRICK GROUNDWATER CHEMISTRY

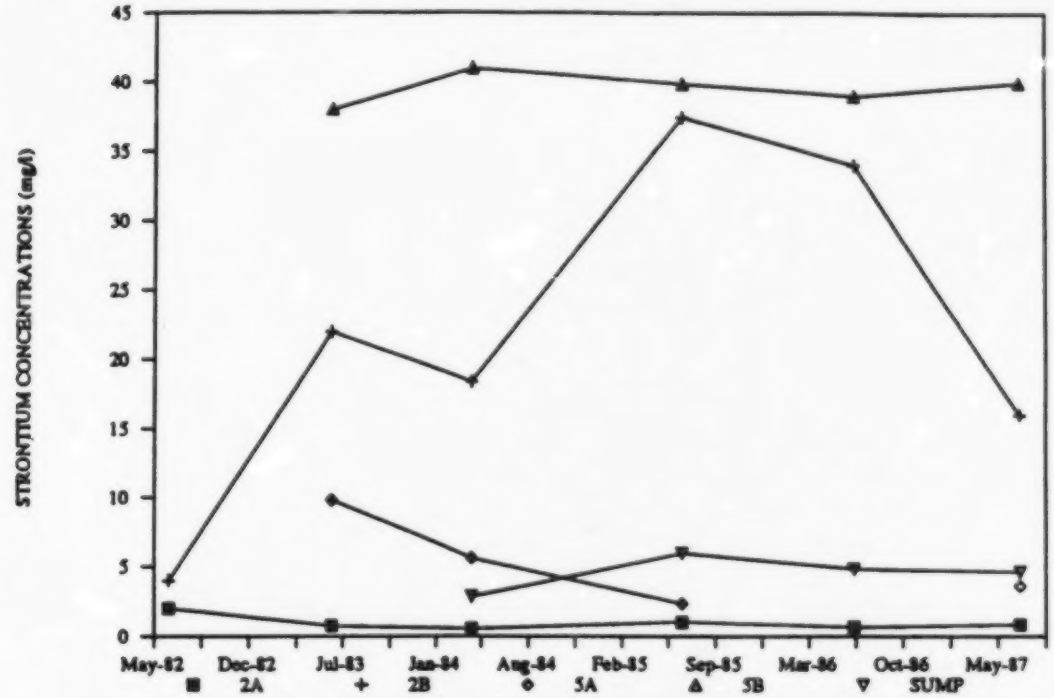


Figure 9a: Strontium levels in upgradient piezometers

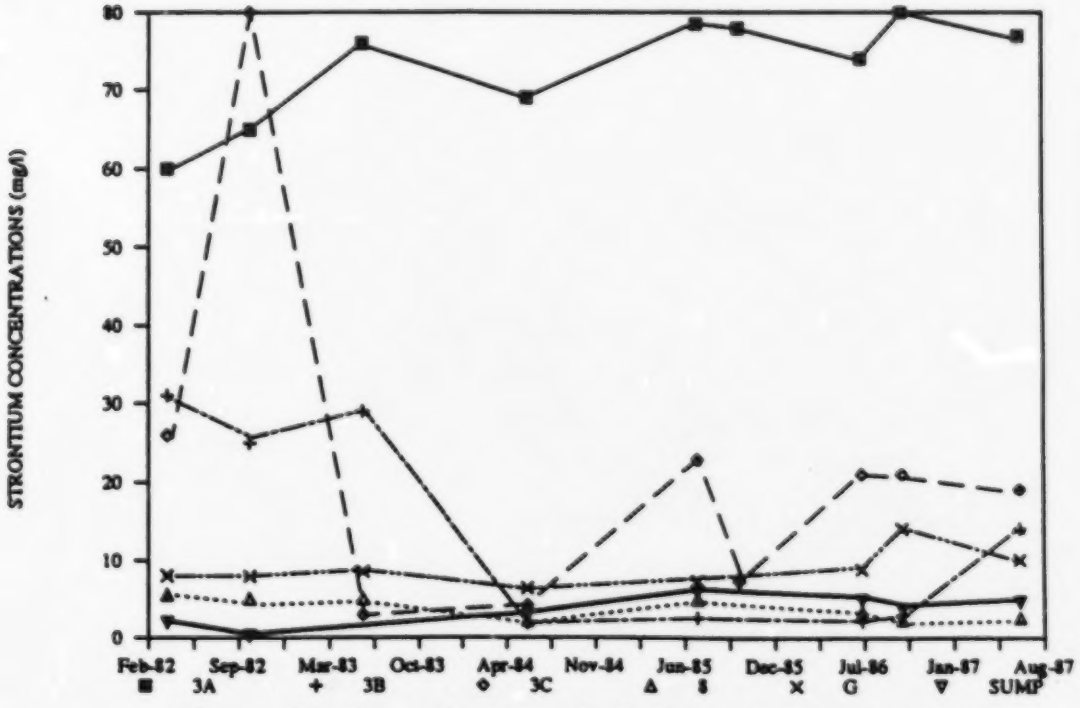
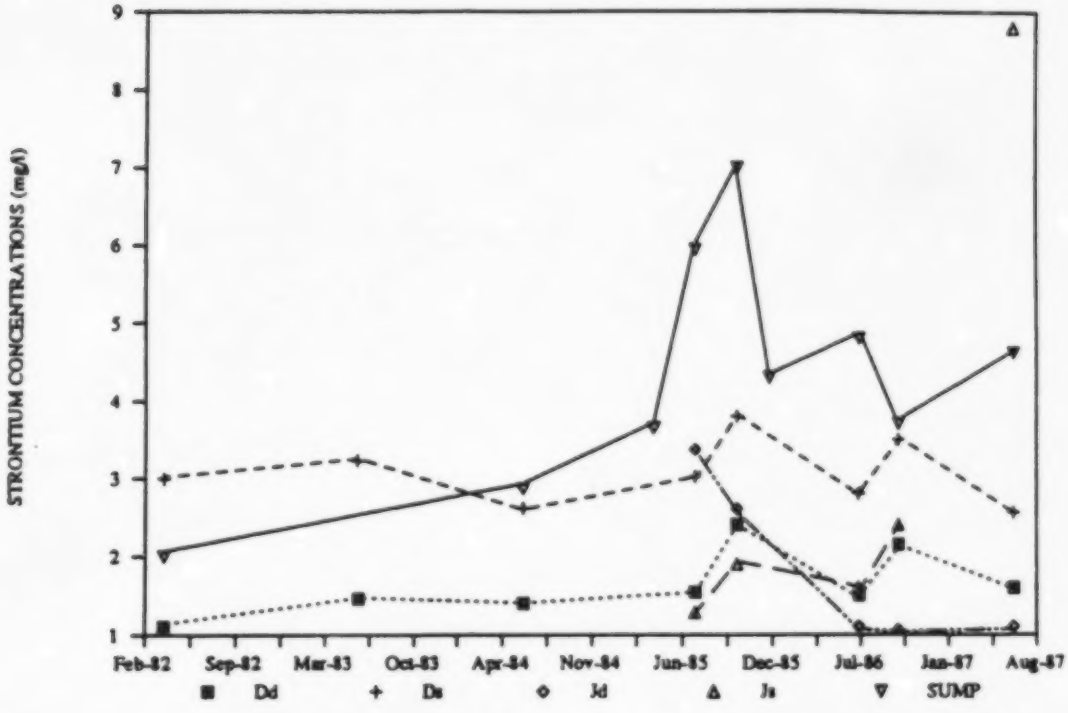


Figure 9b: Strontium levels in "on-site" piezometers

CANADA BRICK GROUNDWATER CHEMISTRY



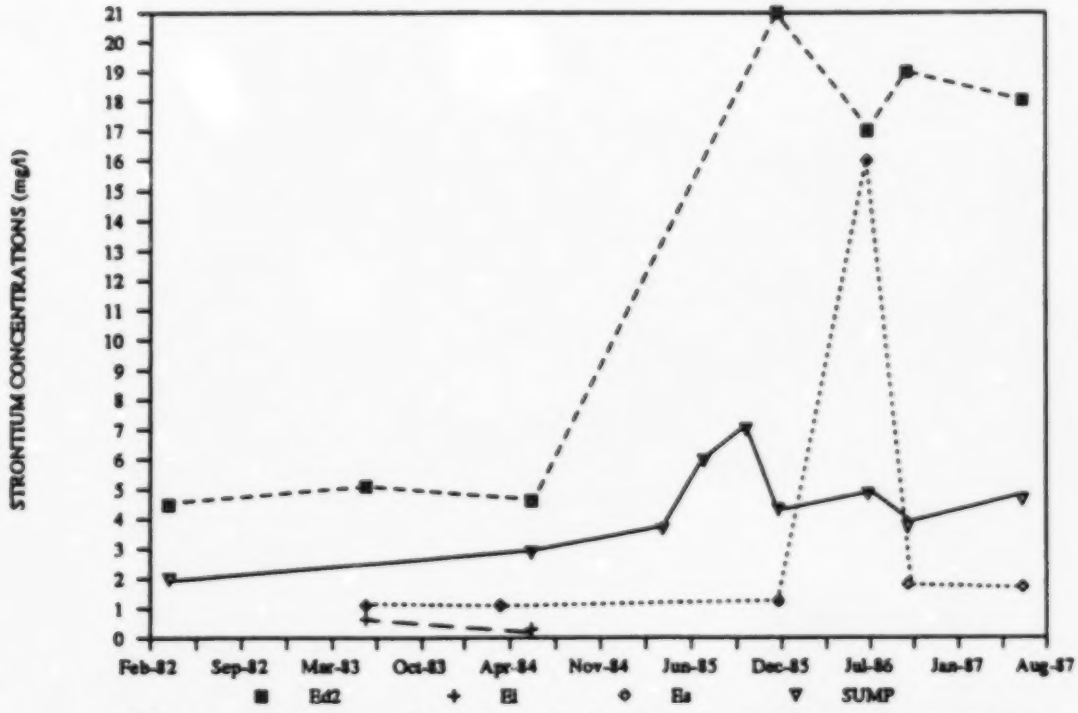


Figure 10 : Strontium levels in downgradient piezometers

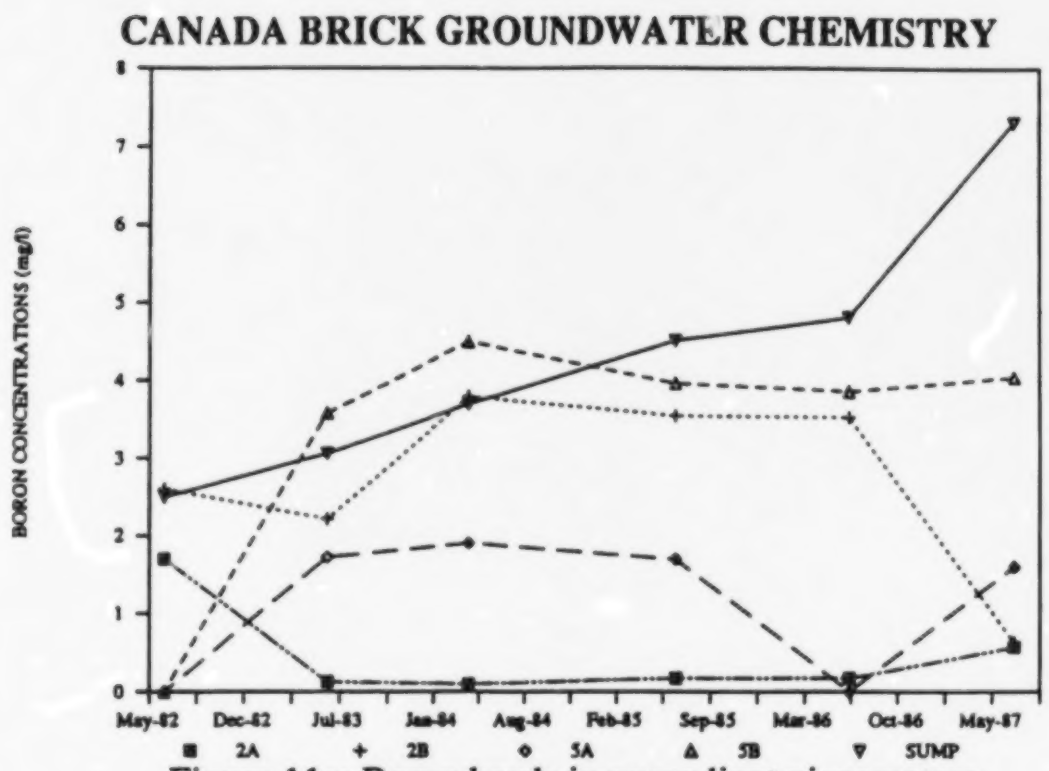


Figure 11a: Boron levels in upgradient piezometers

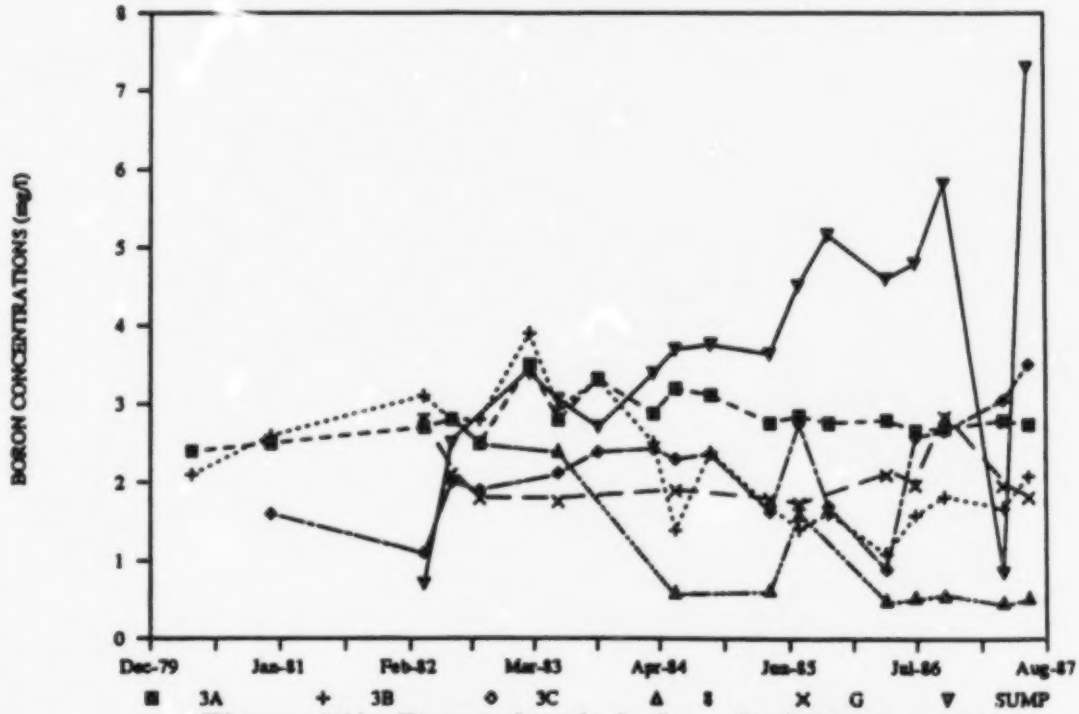


Figure 11b: Boron levels in "on-site" piezometers

CANADA BRICK GROUNDWATER CHEMISTRY

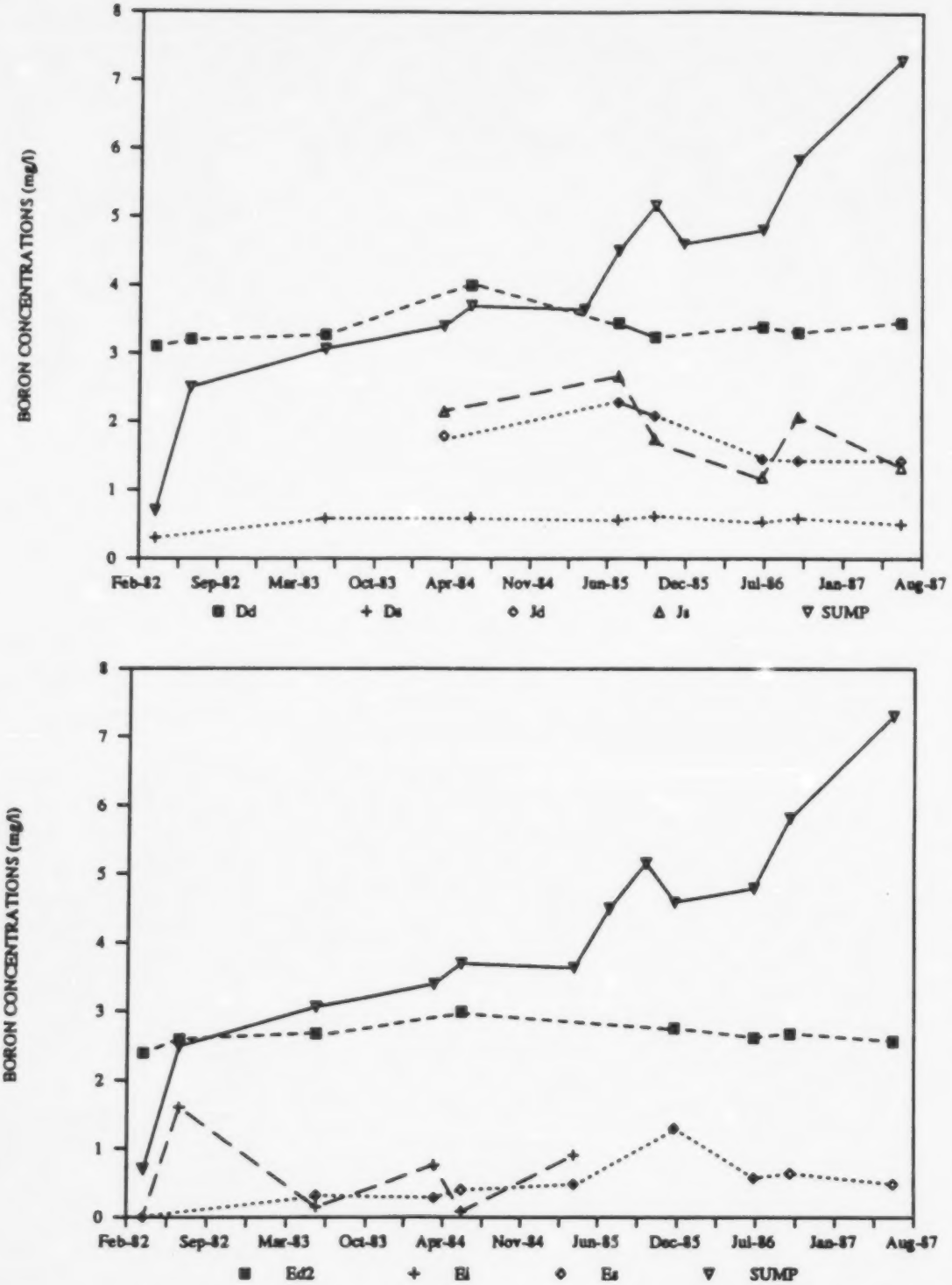


Figure 12 : Boron levels on downgradient piezometers

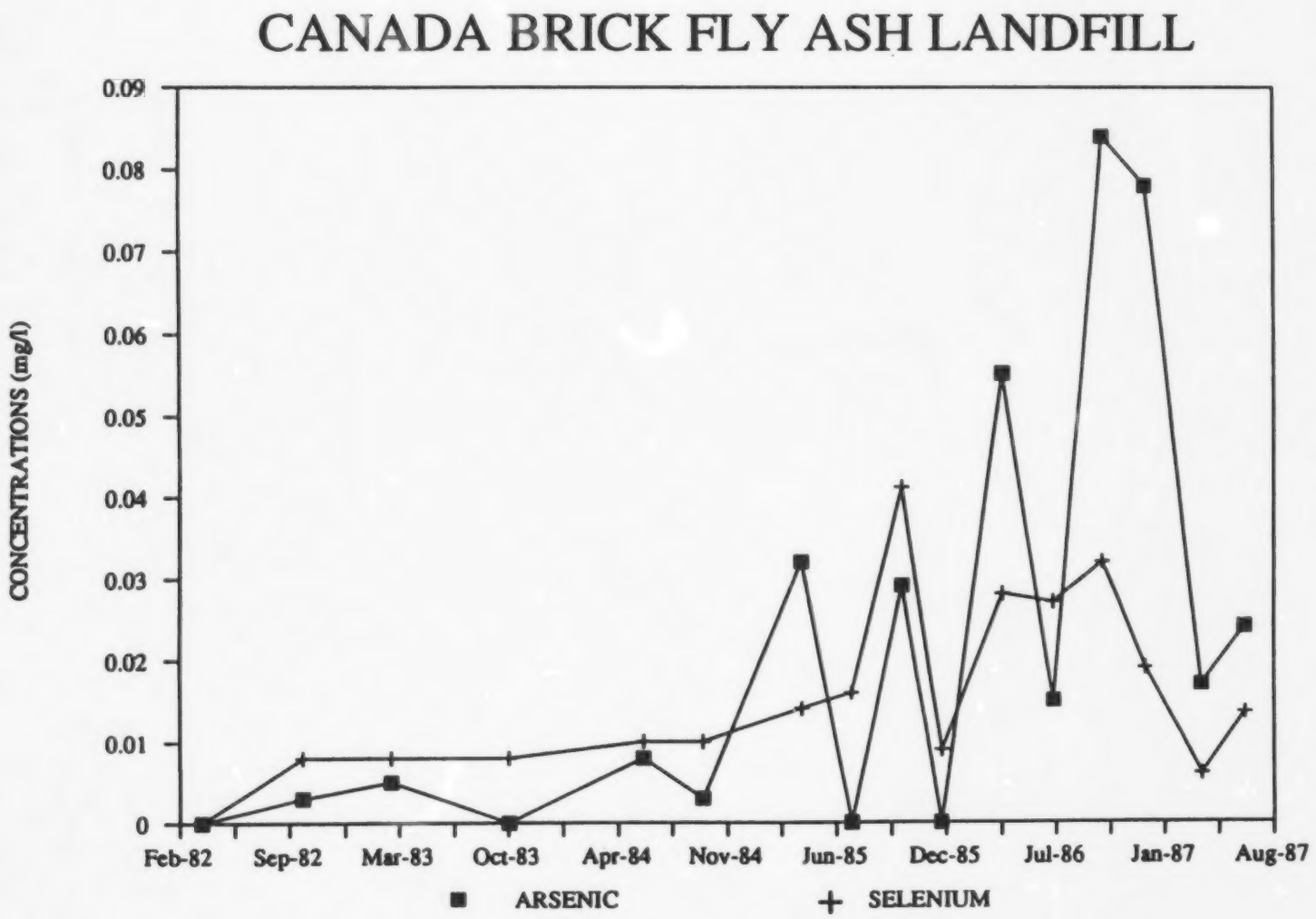


Figure 13 : Sump water trace element chemistry

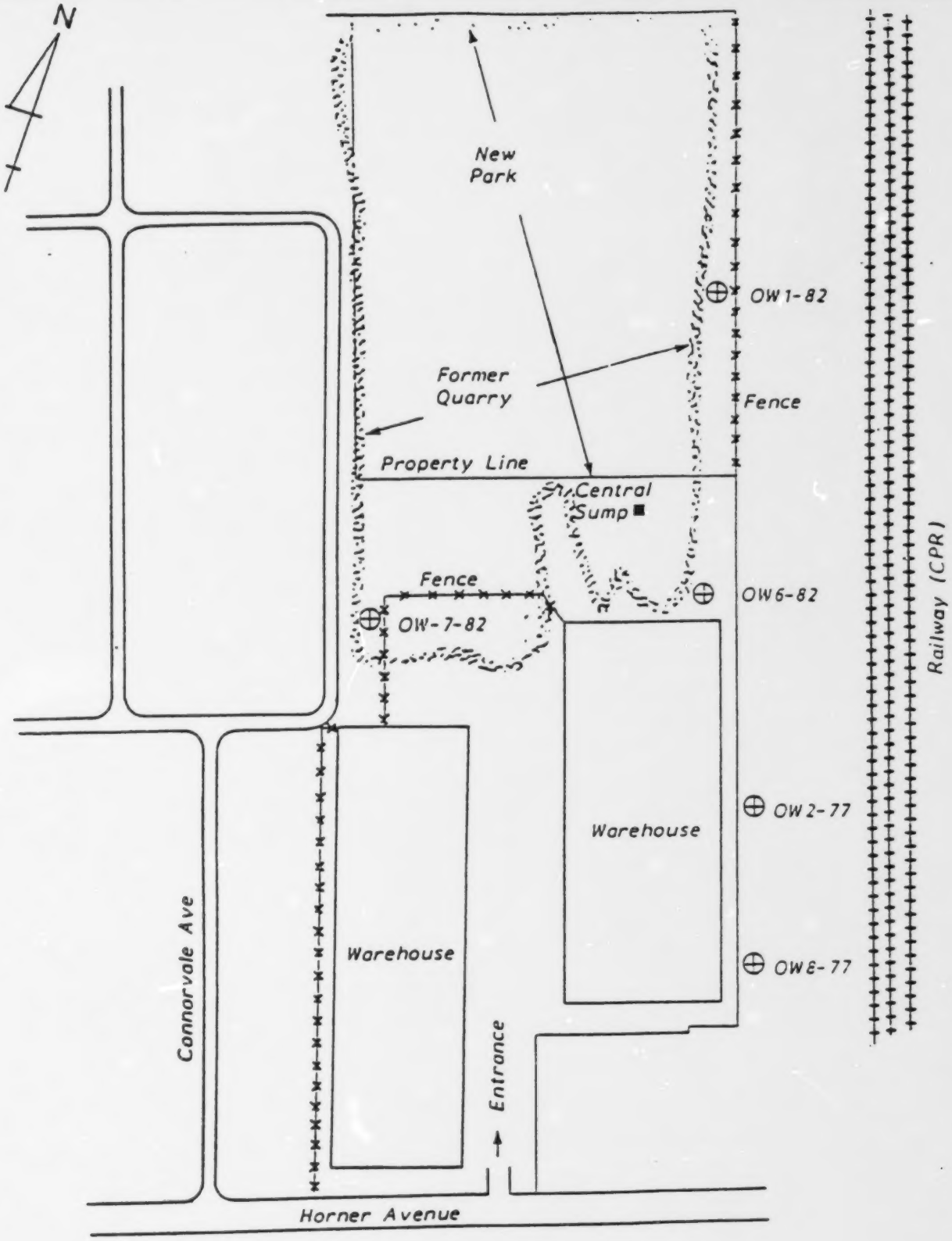


FIGURE 14

BOOTH QUARRY ASH DISPOSAL SITE
(Not to Scale)

158887-RD

BOOTH QUARRY ASH DISPOSAL SITE

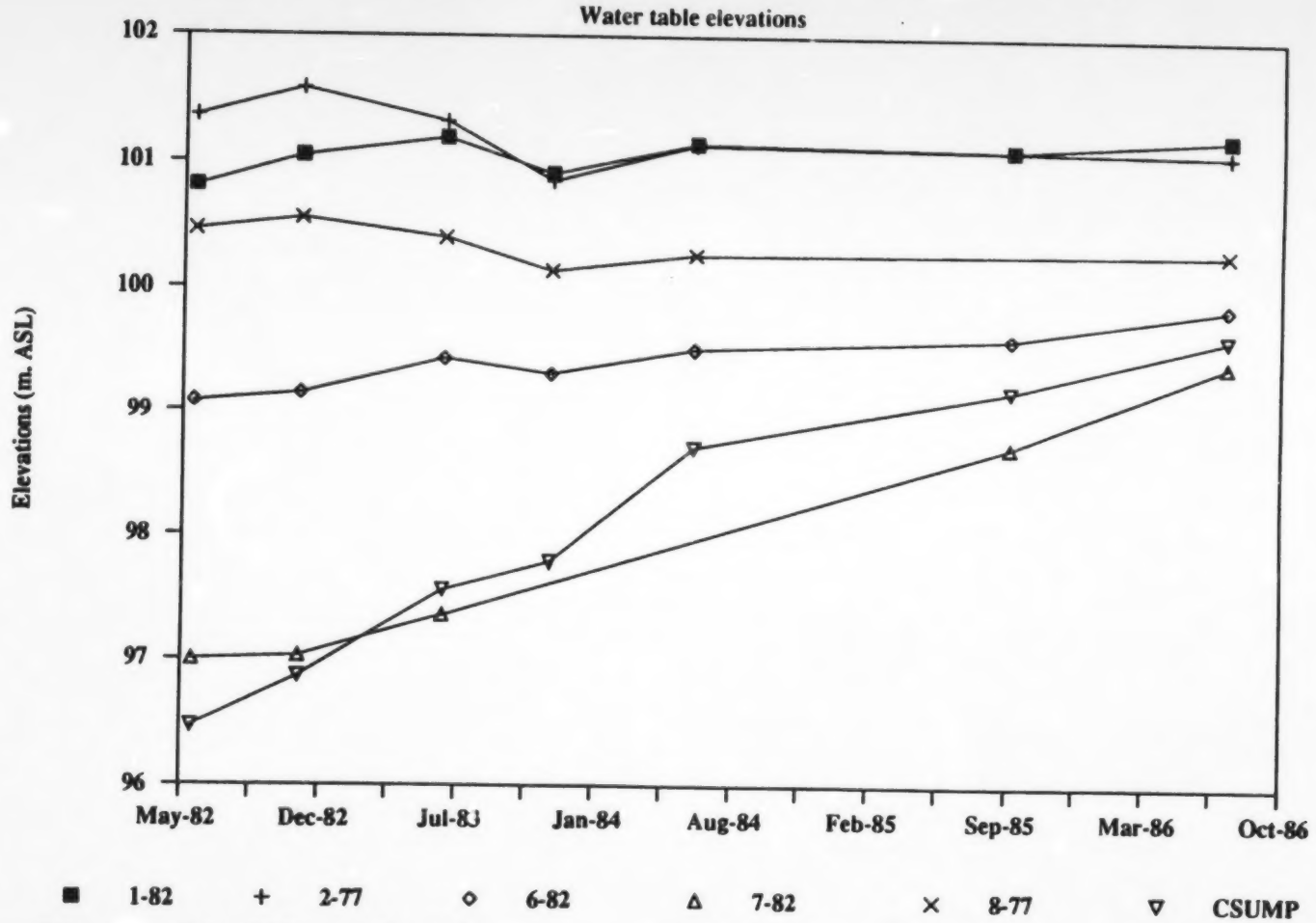
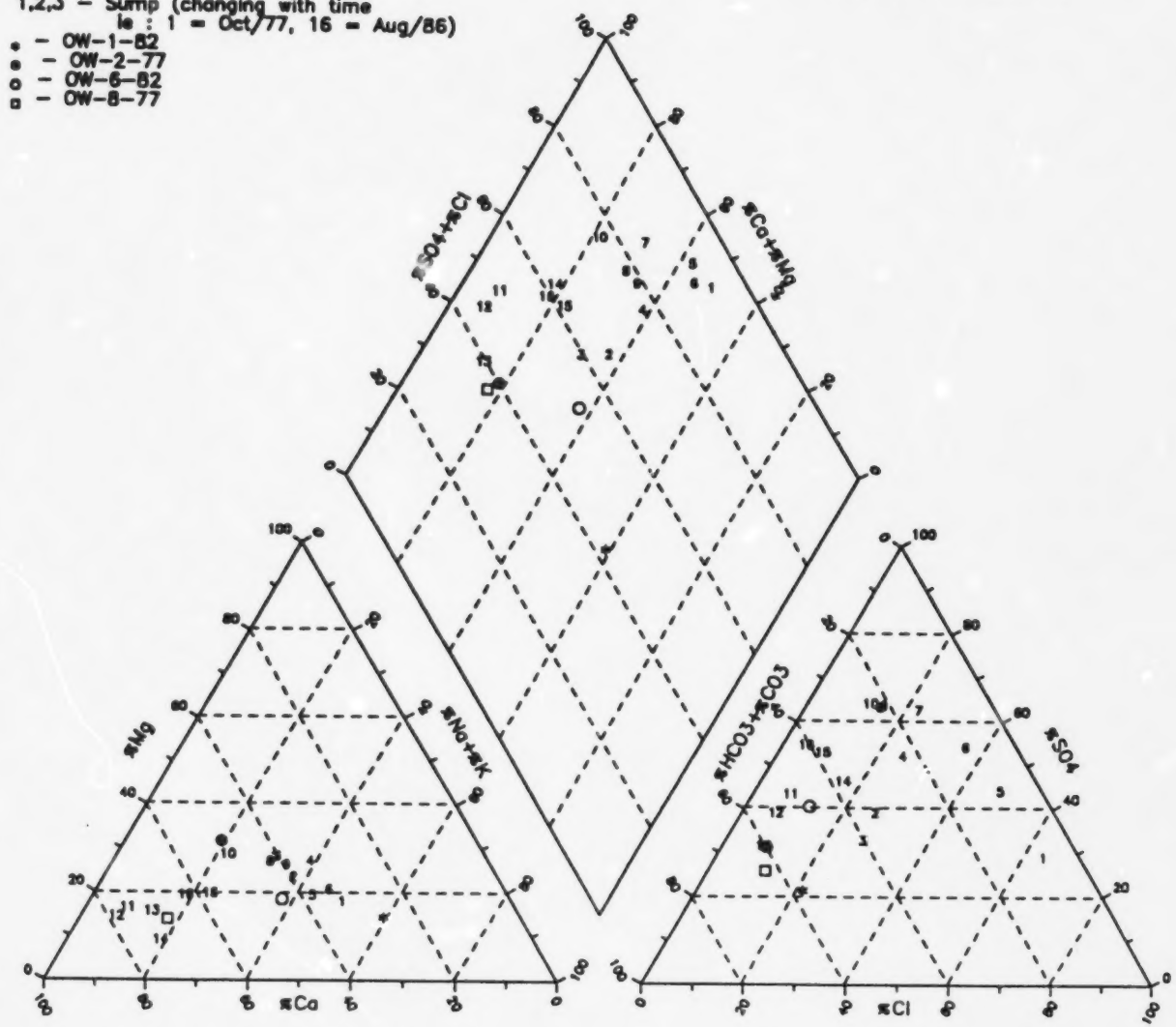


Figure 15: Water level recovery data from sump and piezometers in vicinity of Booth Quarry

PIPER DIAGRAM
MAJOR ION CHEMISTRY

Figure 16 :

BOOTH QUARRY FLY ASH DISPOSAL SITE
- Sump water and piezometer water chemistries
- Monitoring period from June/77 to Aug/86



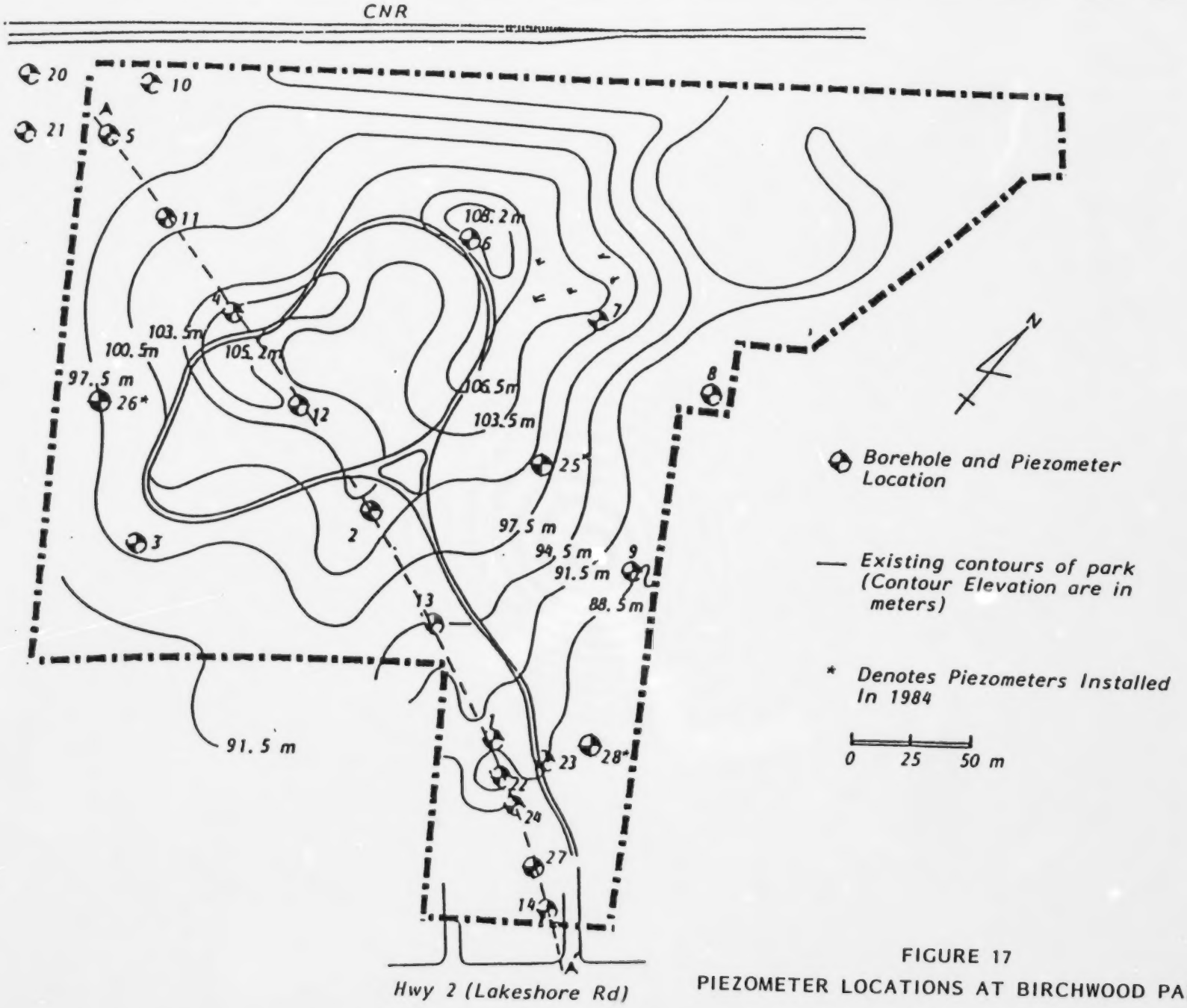


FIGURE 17
PIEZOMETER LOCATIONS AT BIRCHWOOD PARK

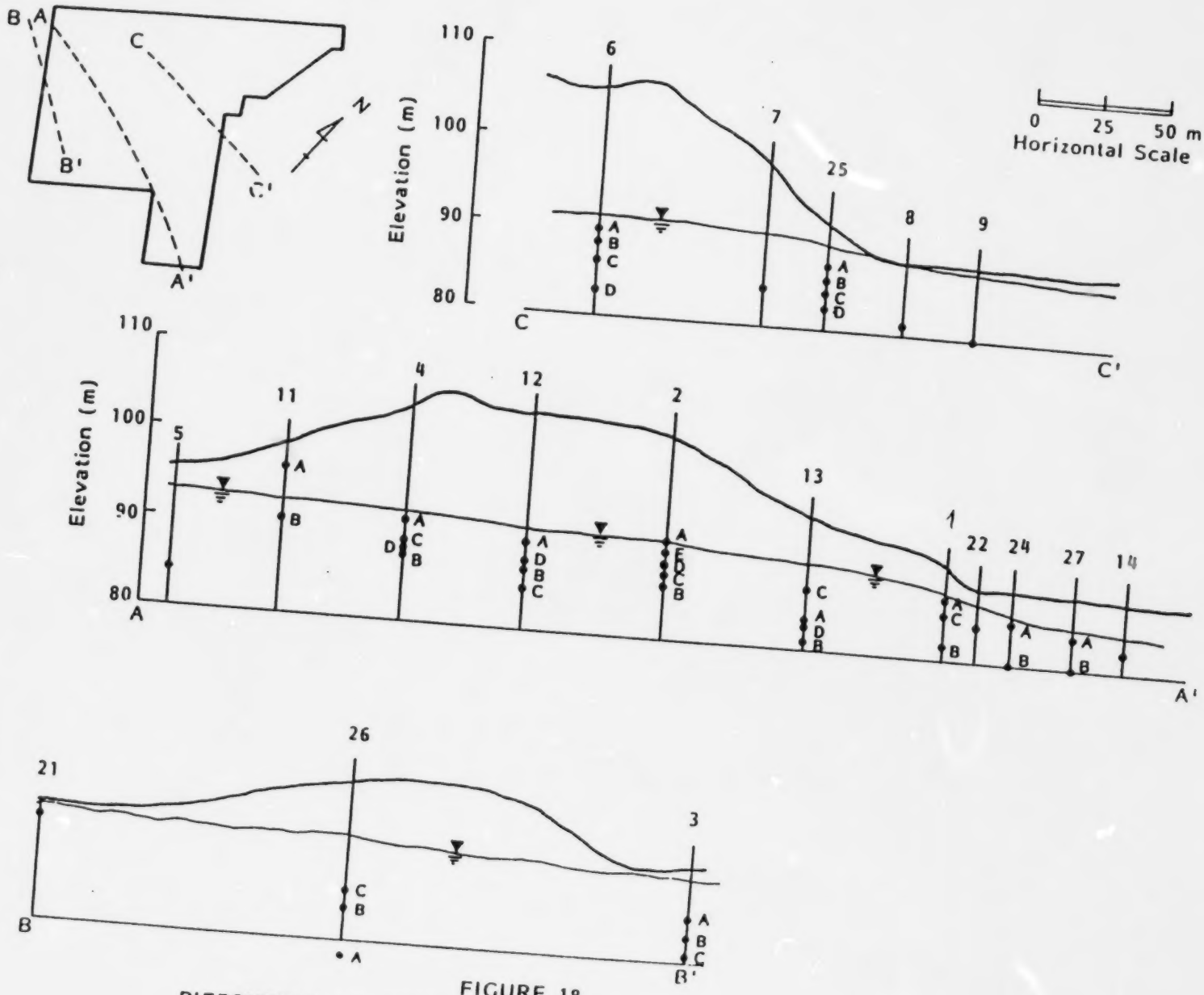


FIGURE 18
PIEZOMETER LOCATIONS AS SHOWN ON THREE SECTIONS

179256-RD
89-42

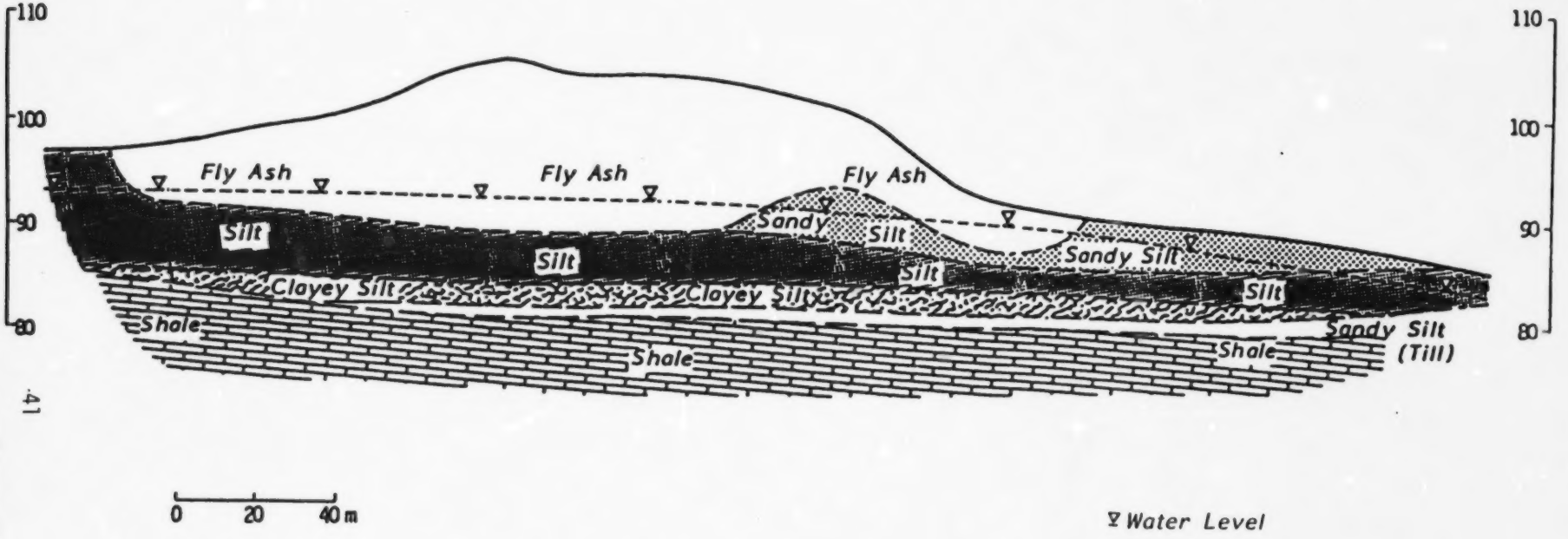


FIGURE 19

STRATIGRAPHY OF BIRCHWOOD PARK ALONG SECTION A-A' AS SHOWN IN FIGURE 2.3a

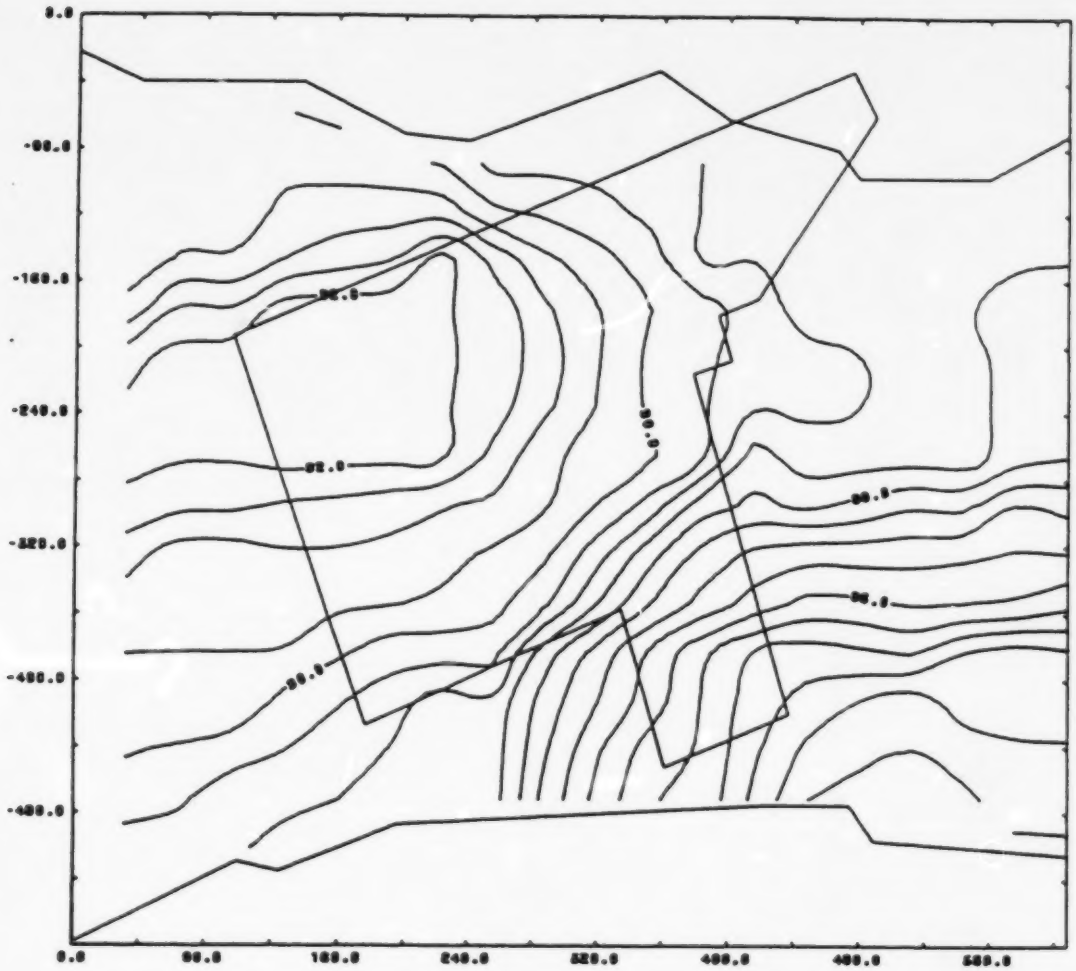


FIGURE 20

KRIGED PHREATIC SURFACE AT BIRCHWOOD PARK

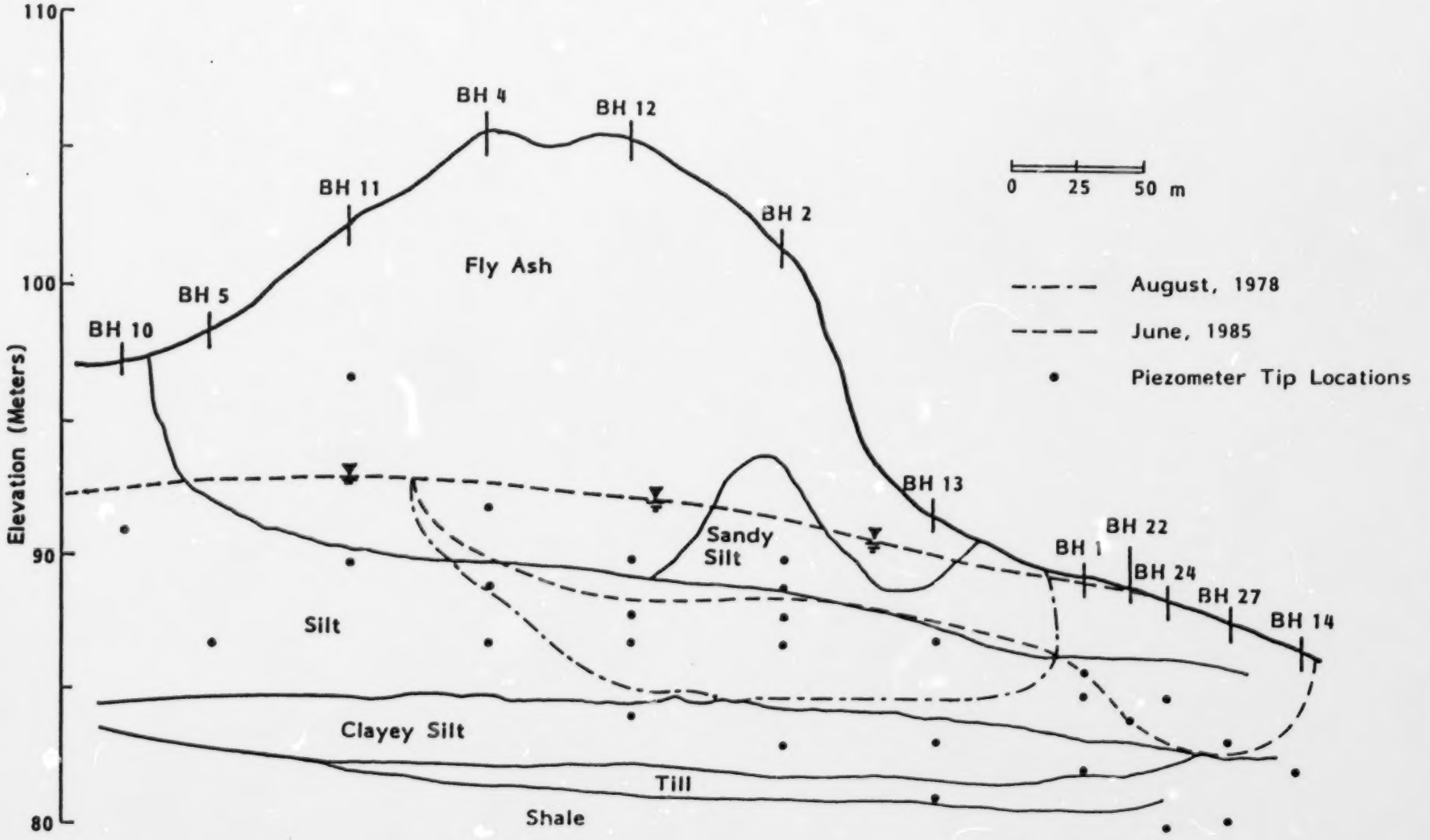


FIGURE 21

BIRCHWOOD PARK, MISSISSAUGA
1000 mg/ℓ SO_4^{2-} CONTOUR LINES

89-42

179259-RD

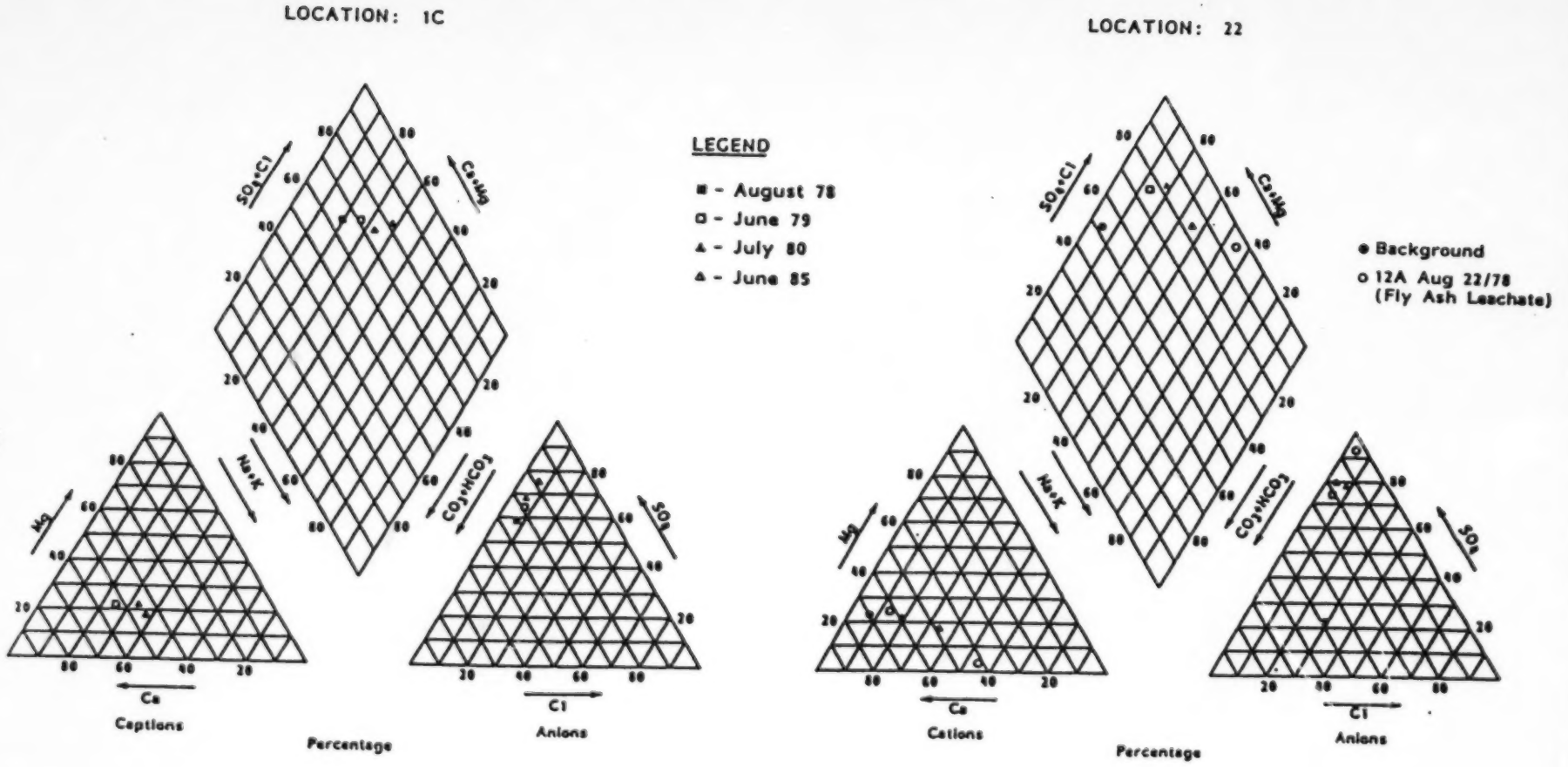


FIGURE 22

PIPER TRILINEAR DIAGRAMS FOR PIEZOMETERS
LOCATED UNDER THE CENTRAL PORTION OF THE ASH FILL